Calcasieu Estuary Remedial Investigation/Feasibility Study (RI/FS): Baseline Ecological Risk Assessment (BERA)

Appendix H2: Assessment of Risks to Carnivorous Wading Birds in the Calcasieu Estuary

Prepared For:

CDM Federal Programs Corporation
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Under Contract To:

Mr. John Meyer, Regional Project Manager
U.S. Environmental Protection Agency, Region 6
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Prepared – October 2002 – By:

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Under Contract To:

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Appendix H2. Assessment of Risks to Carnivorous Wading Birds in the Calcasieu Estuary

1.0 Introduction

Development and industrialization in and around the Calcasieu Estuary in southwestern Louisiana in recent decades has led to concerns of environmental contamination in the area. A Remedial Investigation/Feasibility Study (RI/FS) was commissioned to determine the risks posed by environmental contamination to ecological receptors inhabiting key areas of the Calcasieu Estuary. A Baseline Ecological Risk Assessment (BERA) is required to meet this objective. This Appendix is part of the BERA and is conducted in accordance with the procedures laid out by the USEPA in the *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessment* (USEPA 1997a). Under the eight-step process described by the USEPA for conducting a BERA, a screening ecological risk assessment (SERA) must first be conducted to determine preliminary estimates of exposure and risk.

The SERA for the Calcasieu Estuary (CDM 1999) identified areas of concern (AOCs), contaminants of concern (COCs), and ecological receptors potentially at risk. The SERA findings were revisited in a Baseline Problem Formulation (BPF) to yield a refined list of contaminants of concern, areas of interest, and ecological receptors to be considered in the BERA. The Phase II data collection provided more information and, therefore, a better tool to estimate risk at a screening level. Using this information, a conservative, deterministic assessment was conducted and can be found in Appendix G along with a description of the methods used to identify the COCs and areas of concern for carnivorous wading birds.

This Appendix is organized as follows. Section 1 provides a brief overview of the results of the conservative, deterministic ERA for wildlife described in detail in Appendix G. The AOCs and COCs that screened through the conservative, deterministic assessment for carnivorous wading birds are described in this section. Section 1 also includes a description of the conceptual model for carnivorous wading birds in the Calcasieu Estuary. A statement outlining the purpose of this assessment concludes Section 1.

Section 2 describes the probabilistic risk assessment methods used to estimate risks of COCs to carnivorous wading birds in the Calcasieu AOCs. Section 3 describes the probabilistic risk assessment results and Section 4 identifies the sources of uncertainty that could influence the estimated risks for carnivorous wading birds. The final section of this appendix, Section 5, contains the conclusions regarding risks of COCs to carnivorous wading birds in the Calcasieu Estuary.

1.1 Deterministic Ecological Risk Assessment Summary

The methods and results of the deterministic ecological risk assessment are presented in detail in Appendix G. In summary, the deterministic assessment used a conservative approach to estimate risk to carnivorous wading birds from chemicals of potential concern (COPCs) in the Bayou d'Inde, Upper Calcasieu River, and Middle Calcasieu River Areas of Concern (BI AOC, UCR AOC, MCR AOC, respectively) of the Calcasieu Estuary system. Several reference sites, including Bayou Connine Bois and Choupique Bayou, were also included in the deterministic assessment to provide a basis of comparison of risks. The deterministic assessment compared potentially attainable high exposures with conservative adverse effects benchmarks to provide a means of identifying which chemicals are of potential concern to carnivorous wading birds and in which areas of the Calcasieu Estuary system. Generally, a risk quotient (total daily intake/effect dose) for carnivorous wading birds greater than one for any COPC in any of the Calcasieu areas resulted in the chemical being screened through to the probabilistic ecological risk assessment. COPCs that screened through the SERA are now referred to as COCs. Mercury was screened through in all three areas, whereas TCDD and equivalents (TCDD-TEQs) were screened through in BI AOC only. The reference areas were also screened through to the probabilistic risk assessment so that risks in the AOCs could be compared to background risks. Results of the deterministic risk assessment are presented in Table H2-1.

1.2 Contaminants of Concern

The COCs that screened through include mercury and TCDD-TEQs and are described below.

Mercury

Mercury is found in the environment as the metal, Hg^0, and as divalent mercuric Hg(II) species. In the water column, Hg^0 is oxidized to Hg(II) under acidic conditions. Hg(II) undergoes a number of important reactions, one of which is methylation by microbes and adsorption and absorption by biota (Stein *et al*. 1996). Biomethylation occurs both in the sediments, where sulfate-reducing bacteria are the primary methylators of mercury, and in the water column (Winfrey and Rudd 1990). Methylation in the water column also occurs abiotically, mediated by dissolved organic carbon (Weber 1993). Methylmercury may make up as much as 25 percent of the mercury in rivers and lakes (Gilmour and Henry 1991).

Methylmercury is highly soluble in water, extremely mobile, and thus readily enters the aquatic food web. Because methylation is higher under anaerobic conditions, benthic organisms in the anaerobic zones of sediment may be exposed to high methylmercury concentrations. These organisms are consumed by a variety of species, including sediment-probing birds, leading to biomagnification up the food chain. The accumulation of methylmercury in aquatic organisms has been well documented, with concentrations in carnivorous fish 10,000 to >1,000,000 times the concentrations found in ambient waters (Stein *et al*. 1996). Gilmour and Henry (1991) showed that fish from contaminated systems may continue to contain high levels of methylmercury long after inputs to the systems have ceased. Also, the efficient assimilation of the lipophilic methylmercury in fat and muscle and the lack of elimination results in increasing methylmercury concentrations with the age and size of fish and wildlife predators.

This assessment focuses on the risks posed by methylmercury to sediment-probing birds because this species of mercury is more readily bioaccumulated and more toxic to wildlife than is metallic mercury. Further, previous assessments of methylmercury risks to wildlife have shown that species higher in the aquatic food chain are at particular risk of experiencing adverse effects, including reduced reproduction, impaired growth and development, and death (MacIntosh *et al*. 1994; USEPA 1997b; Moore *et al*. 1999). Sediment-probing birds are fairly high in the food chain and are potentially at high risk of exposure to methylmercury because they consume sediment-dwelling invertebrates as well as sediments via incidental ingestion.

TCDD-TEQs

Tetrachlorinated dibenzo-*p*-dioxins and equivalents represent a group of aromatic compounds with similar properties (WHO 1989). The term equivalents refers to a specific group of polychlorinated dibenzo-*p*-dioxin (PCDDs) congeners, polychlorinated dibenzofuran (PCDFs) congeners and polychlorinated biphenyl (PCB) congeners. This group has a common structural relationship that includes lateral halogenation and the ability to assume a planar conformation. The planar conformation is important as it leads to a common mechanism of action in many animal species that involves binding to the aryl hydrocarbon (Ah) receptor and elicitation of an Ah receptor-mediated biochemical and toxic response (van den Berg *et al.* 1998; Safe 1994).

Each of these compounds, while similar in structure and acting at the same receptor, has different potencies, depending on the individual congener. To address these issues and effectively estimate the relative toxicity of these mixtures, a system has been created involving the development and use of toxic equivalency factors (TEFs). This approach is based on the *in vivo* and *in vitro* toxicity of each of the compounds in relation to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). TCDD is considered to be most toxic member of the this class of chemicals (van den Berg *et al.* 1998; Birnbaum and DeVito 1995; Safe 1994) and the toxicity of the others depends on the degree of chlorination, the chlorination sites, and the ability to achieve a planar form, relative to TCDD. There are a number of assumptions made when using the TEF approach. These include: 1. the congeners are Ah-receptor antagonists and their toxicological potency is mediated by their binding affinity, and 2. no interaction occurs between the congeners and thus the sum of the individual congener effects accounts for the potency of the mixture. The overall effect of these assumptions is a potency estimate or toxic equivalent (TEQ) value. A more detailed discussion of the TEF approach for expressing the toxicity of this class of chemicals is presented in Appendix G.

The environmental degradation and metabolism of the congeners varies due to their unique physical/chemical properties. These can cause substantial differences between the congeners detected in environmental samples and the congener makeup of the original product (van den Berg *et al.* 1998). The majority of these congeners have low solubility, low vapor pressure and high resistance to chemical breakdown, and are, therefore, highly persistent in the environment. They are also highly lipophilic with a high propensity to bind to organic and particulate matter. When released to aquatic systems, the majority of these compounds form associations with dissolved and/or particulate matter in the water column; biodegradation is considered to be a relatively minor fate process in water (NRCC 1981; Howard *et al*. 1991). Aquatic sediments provide a sink for these compounds and may represent long term sources to the aquatic food web (Kuehl *et al*. 1987; Corbet *et al*. 1983; Tsushimoto *et al.* 1982; Muir *et al*. 1985). As sediments are resuspended and carried downstream, they tend to accumulate in areas where currents are slow and the particles have time to settle.

Organisms may be exposed to TCDD-TEQs through trophic transfer. PCDDs, PCDFs and PCB congeners are highly bioaccumulative substances that increase in concentration as they are passed up the food chain (i.e., biomagnification). For organisms inhabiting the Lake St. Clair ecosystem, Haffner *et al*. (1994) observed that PCB concentrations increased from 935 ng/kg in sediments, to 1.36 mg/kg in bivalves, to 7.24 mg/kg in oligochaetes, and to 64.9 mg/kg in predatory gar pike. Mink are particularly sensitive to PCBs and similar chemicals (Moore *et al*. 1999). Research has found that they accumulate PCBs in their subcutaneous fat at levels 38 to 200 times dietary concentrations, depending on the PCB congener (USEPA 1993). The avian predators of the Calcasieu estuary study area would similarly be expected to accumulate PCBs from the prey they consume.

This assessment estimates the risks posed by coplanar congeners to sediment-probing birds because these compounds are expected to biomagnify up the food chain.

Further, previous assessments have shown that species higher in the aquatic food chain are at particular risk of experiencing adverse effects, including reduced reproduction, impaired growth and development, and death (Moore *et al*. 1999; Tillitt *et al*. 1996; Heaton *et al*. 1995). Sediment-probing birds are moderately high in the food chain and are potentially at high risk of exposure to coplanar congeners because they consume invertebrates and sediments.

1.3 Receptors of Concern

Thorough observations of the study area led to the identification of a number of carnivorous wading bird species including blue heron, great egret, white ibis, and roseate spoonbill (ChemRisk 1996). The named species are commonly observed in the study area and are opportunistic feeders that may consume fish and aquatic invertebrates as parts of their diets. The following sections review the life histories and foraging behaviours of these four species. This information is then used to develop the life history and foraging behavior of a hypothetical receptor that will be used in this assessment. The exposure assessment for carnivorous wading birds exposed to COCs will be based on a hypothetical receptor that incorporates many of the characteristics typical of the listed species rather than focussing on any particular species.

Great Blue Heron (*Ardea herodias*)

The great blue heron is a large, long-necked wading bird up to 1.0 m tall with males and females being similar in size and weight. Quinney (1982) and Hartman (1961) report weights ranging between 2.2 and 2.6 kg. Herons prefer habitat that includes wetland/open water characteristics (Ehrlich *et al*. 1988). Their natural range includes the continental USA and southern Canada. Most of the feeding occurs in shoreline areas of lakes, ponds, beaver dams, slow-moving rivers and streams, and estuaries.

Herons primarily feed on fish, but they also consume small quantities of aquatic invertebrates, small mammals, and reptiles and amphibians (Ehrlich *et al*. 1988). Alexander (1977) found that up to 98% of stomach contents were fish. Two percent were identified as crustaceans and amphibians. Quinney (1982) reported that the entire stomach contents consisted of fish.

Herons often travel extensively between nesting grounds and foraging areas. The distances travelled can range from 2.3 to 6.5 km. However, little is known about home range size during the non-breeding period. Presumably, the birds stay close to food sources.

Feeding territory sizes vary depending on the quality of the habitat in terms of food type and availability. The feeding range can span 129 to 355 m of the shoreline or a land area of 0.6 to 8.4 ha.

Great Egret (*Ardea alba*)

Life history information for the great egret is very similar to that for the blue heron. However, the great egret tends to be smaller than the blue heron, reaching a height of about 80 cm (Robbins *et al*. 1966). Males and females have similar morphology, which is characterized by long necks, bills, and legs as well as striking white plumage (Ehrlich *et al*. 1988). Average body weights for egrets are about 0.812 kg for females and 0.935 kg for males (Kushlan 1977). Egrets prefer wetland habitat that also contains open water. This habitat is necessary to provide the birds with plenty of fish and aquatic invertebrates. Egrets also consume reptiles, amphibians, and small mammals, though, in smaller quantities.

White Ibis (*Eudocimus albus*)

White ibis is the smallest species of this guild. Its average body weight is about 0.76 kg for females and 1.04 kg for males (Kushlan 1977). It stands about 50 cm tall and has a characteristic decurved bill (Robbins *et al*. 1966). Its plume is white with black extremities. Like other birds from this guild, it likes to wade in shallow wetland habitats in search of aquatic prey. Prey includes fish primarily, with some invertebrates (Erlich *et al*. 1988).

Roseate Spoonbill (*Ajaia ajaia*)

Roseate spoonbills are large wading birds reaching a height of 80 cm and weight of 1.8 kg (Robbins *et al*. 1966). Their plumage is pink with scarlet highlights on the wings. The birds have spatula-like bills up to 18 cm long. They use this highly adapted tool to probe shallow water for fish and to a lesser extent for invertebrates. These birds prefer secluded habitats in remote swamplands surrounded by dense aquatic and mangrove plants.

Hypothetical Receptor of Concern

The hypothetical carnivorous wading receptor for this assessment has a blend of the life history characteristics and foraging behaviour patterns exhibited by species in this foraging guild. These characteristics are as follows:

- The receptor body weight is approximately equal to the average of the species considered above. The coefficient of variation (CV) for body weight is also approximately equal to the CV for adults of carnivorous wading bird species (i.e., 15%). To ascertain whether smaller carnivorous wading birds might be at greater risk because of their higher metabolic rate when normalized to body weight, we conducted a "what if" analysis using a receptor body weight typical of white ibis;

- The receptor forages on a local scale, has high site fidelity, and is non territorial. Thus, multiple individuals are assumed to forage for long periods of time exclusively within subareas of concern;

- It is assumed that each area has shallow, open or semi-vegetated areas and has mudflats and sandy beaches preferred by carnivorous wading birds for foraging. These habitats are common in the AOCs and reference areas of the Calcasieu Estuary;

- The hypothetical receptor is assumed to be resident year round in the Calcasieu Estuary; and,

- The diet of the hypothetical receptor is assumed to consist entirely of fish.

1.4 Conceptual Model

The conceptual model illustrates the relationships between sources and releases of COCs, their fate and transport, and the pathways through which COCs reach carnivorous wading birds and exert adverse effects. The model enhances the level of understanding regarding the relationships between human activities and ecological receptors at the site under consideration. In so doing, the conceptual model provides a framework for predicting effects on ecological receptors and a template for generating risk questions and testable hypotheses (USEPA 1997b; 1998). The conceptual site model developed for the Calcasieu Estuary is described in greater detail in Chapter 7 of the BPF. It summarizes information on the sources and releases of COCs, the fate and transport of these substances, the pathways by which ecological receptors are exposed to the COCs, and the potential effects of these substances on the ecological receptors that occur in the Calcasieu Estuary. In turn, this information is used to develop a series of risk hypotheses that provide predictions regarding how ecological receptors will be exposed to and respond to the COCs.

Carnivorous wading birds are exposed to a number of chemicals of potential concern (COPCs) in the Calcasieu Estuary system and the deterministic risk assessment (Appendix G) identified those COCs that pose risks to these animals. Specifically, carnivorous wading birds are at greatest risk from mercury and TCDD-TEQs. These contaminants of concern are available for uptake by carnivorous wading birds, primarily through the food chain. The Phase II sampling program provided data identifying substantial residues of these COCs in prey fish. Other routes of exposure, including inhalation, water consumption, and transdermal transfer have been excluded from this assessment as their contribution to overall exposure is likely negligible.

1.5 Assessment Endpoints

An assessment endpoint is an 'explicit expression of the environmental value that is to be protected' (USEPA 1997a). The selection of assessment endpoints is an essential element of the overall ERA process because it focuses assessment activities on the key environmental values (e.g., reproduction of carnivorous wading birds) that could be adversely affected by exposure to environmental contaminants. Assessment endpoints must be selected based on the ecosystems, communities, and species that occur, have historically occurred, or could potentially occur at the site (USEPA 1997a).

A measurement endpoint is defined as 'a measurable ecological characteristic that is related to the valued characteristic selected as the assessment endpoint' and it is a measure of biological effects (e.g., mortality, reproduction, growth; USEPA 1997a). Measurement endpoints are frequently numerical expressions of observations (e.g., toxicity test results, community diversity measures) that may or may not be compared to similar observations at a control and/or reference site.

To support the identification of key assessment and measurement endpoints for the Calcasieu Estuary BERA, the United States Environmental Protection Agency (USEPA) convened a BERA workshop in Lake Charles, LA on September 6 and 7, 2000. The workshop participants included representatives of the USEPA, United States Geological Service (USGS), National Oceanic and Atmospheric Administration (NOAA), Louisiana Department of Environmental Quality (LDEQ), United States Fish and Wildlife Service (USFWS) and CDM Federal. The workshop was designed to enable participants to articulate the goals and objectives for the ecosystem (i.e., based on the input that had been provided by the community in a series of public meetings), to assess the state of the knowledge base, to define key issues and concerns, and to identify the chemicals and areas of potential concern in the study area. This workshop provided a basis for refining the candidate assessment endpoints that had been proposed based on the results of the SERA (CDM 1999). Workshop participants also identified a suite of measurement endpoints that would provide the information needed for evaluating the status of the assessment endpoints (MacDonald *et al*. 2000).

Aquatic-dependent birds are integrally linked to aquatic ecosystems as a result of their reliance on aquatic organisms for food. Due to their reliance on aquatic organisms for food, it is important to evaluate the effects of environmental contaminants on this group of ecological receptors. An assessment endpoint is survival, growth, and reproduction of carnivorous wading birds.

1.6 Measurement Endpoints

A single measurement endpoint will be used to evaluate the risks to carnivorous wading birds. The potential for adverse effects on this foraging guild will be evaluated using fish tissue data. Specifically, the data on the concentrations of contaminants measured in fish (small sedentary species < 15 cm in length with small foraging ranges) such as killifish, minnows, blennies, gobies, and mollies from each area of concern will be used. These data will be compiled by geographic area within the estuary (based on the diet and foraging range of a hypothetical carnivorous wading receptor), incorporated into a daily intake exposure model, and compared to appropriate toxicity values for survival and reproduction of avian species.

1.7 Risk Hypothesis and Questions

The following risk hypothesis was developed to identify the key stressor-effect relationships that will be evaluated in the probabilistic ecological risk assessment:

> Based on the physical-chemical properties (e.g., K_{ow}s) of the bioaccumulative contaminants of concern, the nature of the food web in the Calcasieu Estuary, and the effects that have been documented in laboratory studies, mercury and TCDD-TEQs released into surface waters accumulate in the tissues of prey fish to levels that adversely affect the survival, growth, and/or reproduction of carnivorous wading birds.

To provide a basis for assessing ecological effects, the assessment endpoint must be linked to the measurement endpoint by risk questions. In this study, the investigation to assess the effects of environmental contaminants to carnivorous wading birds were designed to answer the following risk questions:

- Are the levels of contaminants of concern in the tissues of prey species of carnivorous wading birds in the Calcasieu Estuary sufficiently high to adversely affect survival, growth, or reproduction?
- If yes, what are the probabilities of effects of differing magnitude for survival, growth, and/or reproduction of carnivorous wading birds?

The linkages between the assessment endpoint and the measurement endpoints are articulated in greater detail in Table A1-21 of the BPF.

1.8 Purpose of Appendix

The purpose of this assessment is to test the above risk hypothesis by characterizing the risks posed to the carnivorous wading birds associated with exposure to the COCs identified in Appendix G.

2.0 Methods

A step-wise approach was used to assess risks to the piscivorus bird community posed by the COCs in the Calcasieu Estuary. The steps in this process included:

1. Collection, evaluation, and compilation of the relevant data on the concentrations of COCs in prey items in the Calcasieu Estuary;
2. Assessment of exposure of carnivorous wading birds to COCs (Figure H2-1);
3. Assessment of the effects of COCs on carnivorous wading birds (Figure H2-2); and,
4. Characterization of risks to carnivorous wading birds (Figure H2-3).

Each of these steps is described in the following sections of this report. The results of the deterministic assessment were briefly reviewed in Section 1.1. For details of this assessment, see Appendix G.

2.1 Collection, Evaluation, and Compilation of Data

Information on contaminant levels in prey tissues were collected in two phases, termed the Phase I and Phase II sampling programs. The Phase I program results indicated that the detection limits for many of the COPCs in fish tissues were orders of magnitude above corresponding benchmarks. Therefore, the Phase I results for tissues were not always useful in this assessment. The methods used to collect the tissue samples, quantify the levels of COCs, evaluate the reliability of the data, and compile the information in a form that would support the BERA are described in the following sections.

Sample Collection of Tissues **-** More than 600 tissue samples were collected at sites located throughout the estuary between October, 2001 and November, 2001. Biota tissue samples were collected in three AOCs in the estuary (upper and mid Calcasieu River, and Bayou d'Inde) and in the reference areas (Bayou Connine Bois, Calcasieu Lake, Choupique Bayou, Grand Bayou, and Grand Bayou and Wetlands). There were also a number of sub-areas within the AOCs from which samples were taken. The USEPA Region V FIELDS tools were used to randomly select coordinates (i.e., latitude and longitude) for the assigned number of primary sampling stations and alternate sampling stations (i.e., which were sampled when it was not possible to obtain samples from the primary sampling stations). In the field, each sampling station was located with the aid of navigation charts and a Trimble differentially-corrected global positioning system (GPS). Using standard statistical power analysis methods, an evaluation of previously collected data was completed to determine the number of samples to be collected within each area and sub-area.

The methods used to collect, handle, and transport the tissue samples are described in CDM (2000a; 2000b; 2000c; 2000d; 2000e). Briefly, fish and invertebrate species were collected by hook and line, hand collection and netting. Minnows and other small bait species were collected using legal cast nets, minnow traps, dip nets and bait seines in accordance with the Louisiana Department of Wildlife and Fisheries. Each sample was wrapped in aluminum and put in a Ziploc® bag. All samples were kept frozen and shipped to laboratories in coolers on dry ice.

Chemical Analyses of Tissues - Chemical analysis of the tissue samples was conducted at various contract laboratory program (CLP) and subcontract (non-CLP) analytical laboratories, including USEPA Region VI Laboratory, USEPA Region VI CLP laboratories, Olin Contract laboratories, Texas A&M University laboratories, ALTA laboratories, AATS laboratories and EnChem laboratories. Upon receipt at the laboratory, tissue samples were held in freezers until analysis.

All tissue samples were analyzed for total target analyte list (TAL) metals, target compound list (TCL) semi-volatile organic compounds (SVOCs) and TCL pesticides. Total metals were quantified using the SW6010B method. Polycyclic aromatic hydrocarbons and/or other semi-volatile organic compounds were quantified using the SW8270C method. Methods SW8081A and SW8082 were used to quantify pesticides. Twenty percent of the tissue samples were analyzed for PCB congeners and dioxins/furans. EPA Method SW1668 was used to quantify PCB congeners and SW8290 was used for dioxins/furans.

EnChem laboratories used additional analytical methods to quantify mercury, polycyclic aromatic hydrocarbons (PAHs), pesticides and dioxins and furans. Methods 1631MOD and 1630MOD were used to quantify mercury and methylmercury, respectively. PAHs were quantified using Method 8270C-SIM. Method SW8082 and AXYS Method CL-T-1668A/Ver. 3 were used to quantify pesticides. Dioxins and furans were quantified using AXYS Method DX-T-8290/Ver. 2.

Data Validation and Verification **-** All of the data sets generated during the course of the study were critically reviewed to determine their applicability to the assessment of risks to the biotic community in the Calcasieu Estuary. The first step in this process involved validation of the tissue chemistry data. Following translation of these data into database format, the validated data were then further evaluated to ensure the quality of the data used in the risk assessment**.** We were unable to confirm tissue data results against the original source.

Database Development **-** To support the compilation and subsequent analysis of the information on biota in the Calcasieu Estuary, a relational project database was developed in MS Access format. All of the tissue chemistry data compiled in the database were georeferenced to facilitate mapping and spatial analysis using geographic information system (GIS)-based applications (i.e., ESRI's ArcView and Spatial Analyst programs). The database structure made it possible to retrieve data in several ways, including by data type (i.e., chemistry vs. toxicity), by stream reach (i.e., Upper Bayou d'Inde vs. Lower Bayou d'Inde), by sub-reach (i.e., Upper Bayou d'Inde-1 vs. Upper Bayou d'Inde-2), and by date (i.e., Phase I vs. Phase II). As such, the database facilitated a variety of data analyses.

2.2 Probabilistic Ecological Risk Assessment

Monte Carlo analysis is an increasingly widely used approach to probabilistic risk assessment (USEPA 1997c). It is used to propagate uncertainty associated with the variability of input variables, as well as any incertitude associated with how to parameterize input distributions. In this assessment, we also use probability bounds analysis to determine the relative contributions of incertitude and variability to exposure estimates (see Chapter 9 of MacDonald *et al.* 2001 for more information on the uncertainty analysis approaches used here).

Monte Carlo analysis requires the specification of the statistical distributions of each of the input variables and their interdependencies as measured by correlations. Computer software such as Crystal Ball is used to 'sample' from these distributions and, via the exposure model equation, compute an exposure distribution. This process is repeated many times so as to build up a histogram that serves as the estimate of the full distribution of exposures (explicitly including the tail risks of extreme exposure).

Probability bounds analysis is an exact numerical approach (not based on simulation) that takes as input the same probability distributions used in Monte Carlo simulation, or, when they are difficult to specify precisely, bounds on these distributions (Ferson *et al*. 2002). The method then rigorously computes bounds on the cumulative distribution function. The spread between the bounds of an input or output distribution corresponds directly to the amount of incertitude we have about how to describe the variable. Probability bounds analysis is also useful when independence assumptions are untenable (such as between concentrations in sediments and benthic invertebrates), or when sparse empirical data make it difficult to quantify the correlations among variables.

2.2.1 Exposure Characterization

We estimate exposure of carnivorous wading birds to methylmercury and TCDD-TEQs via a daily intake model. The exposure model calculates the total daily intake of COCs associated with the ingestion of prey fish. Carnivorous wading birds are unlikely to use the saline waters of the estuary as a source of drinking water and the inhalation route of exposure has been shown to be an insignificant source of hydrophobic contaminants in previous assessments of the risks of these substances to aquatic-dependent wildlife (e.g., Moore *et al*. 1999). Chemical assimilation efficiency terms are not included in the exposure equation because the efficiencies of chemical adsorption in wild animals following ingestion will likely be similar to the efficiencies in laboratory animals in toxicity studies. Thus, the chemical assimilation efficiency terms will cancel out when the exposure and effect estimates are combined to estimate risk.

The temporal scale for this assessment is long term because: (1) levels of mercury and TCDD-TEQs are unlikely to exhibit high temporal variability, and (2) chronic toxicity occurs generally at lower levels than acute toxicity. The spatial scale of this assessment is considered to be consistent with home ranges reported for carnivorous wading birds. The foraging area for the hypothetical receptor is set to 2,500 m^2. This area is equivalent to a circular zone of about 52 metres in diameter or 242 metres of shoreline, both of which easily fit into each subarea of interest. This exposure assessment assumes that the hypothetical receptor is present year round in each of the identified Areas of Concern or reference areas.

The exposure model is:

EQUATION #1

$$TDI = \frac{FMR \times C_f}{AE_f \times GE_f}$$

where:

- TDI = total daily intake of COC (mg/kg bw/day),
- C_f = concentration of COC in fish (mg/kg),
- FMR = normalized free metabolic rate (Kcal/kg bw/day),
- GE_f = gross energy of fish (Kcal/kg prey), and
- AE_f = assimilation efficiency of fish (unitless).

Each input variable is described in detail below, including the parameterizations for the Monte Carlo analysis and the probability bounds analysis.

2.2.1.1 Selection of Criteria for Input Distributions

The distributions and distribution parameters used in the exposure analyses are summarized in Table H2-2. Input distributions were assigned as follows: lognormal distributions for variables that are positively skewed with a lower bound of zero and no upper bound (e.g., tissue concentrations), beta distributions for variables bounded by zero and one (e.g., prey assimilation efficiency), and normal distributions for variables that are symmetric and not bounded by one (e.g., body weight). The lognormal distribution is often used to provide good representations for physical quantities that are constrained to being non-negative, and that are positively skewed, such as contaminant concentrations, stream flows, or magnitudes of accidents (Small 1990). Ott (1995) provides an extensive discussion of the theoretical reasons for why contaminant concentrations in the environment are expected to be lognormally distributed. The beta distribution provides a flexible means of representing variability over a fixed range, such as zero to one (Small 1990). The beta distribution can take on a wide variety of shapes between the fixed endpoints and this flexibility has led to its empirical use in diverse applications. The normal distribution arises in many cases because of the central limit theorem, which results in a normal distribution for additive quantities such as body weights (Small 1990). The normal distribution can often be used even for variables that are non negative, as long as coefficients of variation (CV) are small. This is because many distributions converge to a normal distribution as CVs become small. With most random number generators, it is impossible to obtain numbers more than five standard deviations from the mean. Thus, as long as the CV is less than 0.2, there is no concern for selecting negative values for non-negative variables.

2.2.1.2 Input Distributions

Body Weight (Bw)

Body weight data are required in allometric models used to estimate the free metabolic rate. For this assessment, we used body weights that represent average-sized and small carnivorous wading birds.

For the Monte Carlo analysis, the average body weight of herons, great egrets, white ibis, and spoonbills was used (i.e., 1.5 kg). Because the feeding guild encompasses species with widely varying body weights, the calculation of the standard deviation of the mean body weight would have yielded an unduly wide distribution. Instead, we adopted a coefficient of variability (CV) of 15%, which is typical of body weight data for birds. The application of the adopted CV yielded a standard deviation of 0.23.

We also repeated the Monte Carlo analysis with a mean body weight of 0.9 kg (standard deviation equal to 0.14). This body weight is representative of the smallest bird in the guild, the white ibis. Small birds tend to have higher metabolic rates (when normalized to body weight) and, as a result, may be at higher risk of exposure.

Body weights were assumed to be distributed normally. The entire proportion of uncertainty in this variable is likely due to variability, with little incertitude. Thus, probability bounds were not established for this input variable.

Free Metabolic Rate (FMR)

To estimate free metabolic rate, the allometric equation derived by Nagy (1987) was used:

$$FMR = a \cdot BW(g)^b \qquad \textbf{EQUATION \#2}$$

A probabilistic approach was used to estimate *FMR* in both the Monte Carlo and probability bounds analyses, wherein distributions were derived for each of the input variables [body weight (*BW*, *a*, *b)*] and combined according to the above equation. The slope (*a*) and power term (*b*) distributions were based on the error statistics reported in Nagy (1987), assuming an underlying normal distribution for each. For piscivorus birds, log *a* had a reported mean of 0.681 and a standard error of 0.102, and *b* had a reported mean of 0.749 and a standard error of 0.037 (Nagy 1987). The body weight (*BW*) distribution was described above.

Gross Energy of Fish (GE_f)

Gross energies of fish, the main dietary item of carnivorous wading birds, were available from literature. The gross energy of fish is equal to 1,200 Kcal/kg (standard deviation = 240; Thayer *et al*. 1973). The distribution for this variable was assumed to be lognormal. Incertitude was considered low for this input variable because: (1) sufficient experimental data were available to confidently estimate the mean and standard deviation, (2) the variable is easily measured and thus measurement error is low, and (3) there appears to be little difference in the gross energies of different invertebrate species. Therefore, probability bounds were not derived for this variable.

Assimilation Efficiency of Fish (AE_f)

Assimilation efficiencies of waterfowl consuming aquatic prey were studied by Karasov (1990). That study indicated a mean assimilation efficiency of 77% with a standard deviation of 8.4. A beta distribution was assumed for this variable with the following parameterization: alpha = 20, beta = 6.5, and scale = 1.0. This parameterization results in a distribution that has a mean close to 78%. With this distribution, there is approximately a 95% probability that assimilation efficiency will be between 58 and 90%, which would be expected given a standard deviation of 8.4 and a slightly left-skewed distribution. As with gross energy, assimilation efficiency is easily measured and several studies indicated that efficiencies vary little between different bird species consuming fish (USEPA 1993). Therefore, probability bounds were not derived for this variable.

Concentration of COCs in Fish (C_f)

Our exposure model is set to include fish as the only dietary item. This reflects the strong foraging preference of carnivorous wading birds for fish. Although other food items such as aquatic invertebrates might also be opportunistically taken, the relative importance of other foods is low and, therefore, are not included in the exposure model.

Mercury

Concentrations of methylmercury were obtained from field-collected group 1 fish from selected areas of interest. Data for BI AOC included 138 observations with a mean concentration of 0.13 mg/kg and a standard deviation (SD) of 0.132. Mean fish residue for MCR AOC was 0.051 mg/kg (SD= 0.025; n=58), for UCR AOC was 0.041 mg/kg (SD=0.032; n=83), and for the reference areas was 0.023 mg/kg (SD=0.014; n=47).

Because carnivorous wading birds are likely to spatially average their exposures over long durations, we used a bootstrapping approach to estimate mean daily residues in fish over 160 feeding days. The bootstrapping included 160 outer loops (days) and 1,000 inner loops (number of Monte Carlo samples). The resulting means and standard deviations were: 0.130 mg/kg and 0.005 for BI AOC, 0.051 mg/kg and 0.001 for MCR AOC, 0.041 mg/kg and 0.001 for UCR AOC, and 0.023 mg/kg and 0.0001 for the reference areas.

The bootstrap means and standard deviations were used in the Monte Carlo exposure analysis. A lognormal distribution was assumed for these variables. For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H2-3.

TCDD-TEQs

Concentrations of 2,3,7,8-TCDD equivalents (TEQs) in BI AOC fish had a mean of 64.7 ng/kg with a standard deviation of 120 (n=31). Residue levels in reference areas fish averaged 19.1 ng/kg (SD=21.3; n=6). Bootstrap means and standard deviations for BI AOC and the reference areas were as follows: 64.8 ng/kg and 1.15, and 19.1 ng/kg and 0.21, respectively. A lognormal distribution was assumed for this parameter. For the probability bounds analyses, minimum and maximum values were used to parameterize a uniform distribution. The execution of Land statistic was not possible due to the high coefficient of variation associates with these data. The values are listed in Table H2-3.

2.2.1.3 Monte Carlo Analysis

The Monte Carlo analyses for exposure combined the input distributions as specified in Equation 1. Each analysis included 10,000 trials and Latin Hypercube Sampling to ensure adequate sampling from all portions of the input distributions. The analyses were done in Crystal Ball® 2000 (Decisioneering 2000). Considering all possible pairwise combinations of input variables, no dependencies were expected. Therefore, no correlations were included in the Monte Carlo analyses. The Monte Carlo analyses made no distinction in the way incertitude and variability were propagated; they were simply combined. We address incertitude and variability separately in the probability bounds analyses described below.

2.2.1.4 Probability Bounds Analysis

For the probability bounds analyses, we used the same input variables as was used for the Monte Carlo analyses (Table H2-2), with one exception. The exception was in regard to the incertitude arising from small sample size for the concentration variables. The Land statistic was used to develop distributions for mean concentration that reflect our inability to precisely specify the mean because of small sample size.

The remaining input variables were the same as used in the Monte Carlo analyses, reflecting the reasonably good state of empirical knowledge for these variables. The probability bounds analyses were run using RiskCalc, version 4.0.

2.2.2 Effects Characterization

The purpose of this section is to: (1) briefly review the literature on the effects of COCs to avian species, and (2) select the appropriate effects metric that will be used with the results of the exposure assessment to estimate risk. We will focus on ecologically-relevant effects such as survival, growth, and reproduction. Examples of carnivorous wading bird species considered in this section include blue heron, great egret, white ibis, and spoonbill. Because the available toxicological information for carnivorous wading birds is very limited, the literature available for other species of birds and the results of these studies will be discussed where appropriate. Other information on the toxicity of contaminants of concern to wildlife can be found in Appendix 5 of the problem formulation document (MacDonald *et al.* 2001).

Effects data can be characterized and summarized in a variety of ways ranging from benchmarks designed to be protective of most or all species to dose-response curves for the receptor group of interest (e.g., carnivorous wading birds). In this assessment, effects characterization preferentially relies on dose-response curves, but defaults to benchmarks or other estimates of effect [e.g., no observed adverse effect level

(NOAEL), lowest observed adverse effect level (LOAEL)] will be made when insufficient data are available to derive dose-response curves.

The following is hierarchy of decision criteria was used to characterize effects for each COC:

1. Had bioassays with five or more treatments been conducted on the receptor group of interest or a reasonable surrogate? If yes, we estimated the dose-response relationship using the Generalized Linear Model (GLiM) framework described in Kerr and Meador (1996) and Bailer and Oris (1997). The GLiM framework involves conducting linear regression analysis on dose-response data that have been transformed to linearize the relationship (*e.g.*, probit transformation for survival data). If not, we proceeded to 2.

2. Were multiple bioassays available that, when combined, had five or more treatments on the receptor group of interest or a reasonable surrogate? Such bioassays would be expected to have had similar protocols, exposure scenarios and effects metrics. If yes, we estimated the dose-response relationship as in 1. If not, we proceeded to 3.

3. Had bioassays with less than five treatments been conducted on the receptor group of interest or a reasonable surrogate? If yes, we conducted hypothesis testing to determine the NOAEL and LOAEL or reported these metrics when available from the original study. If not, we proceeded to 4.

4. Were sufficient data available from field studies and monitoring programs to estimate concentrations or doses of COCs consistently associated with

no adverse effects and with adverse effects to carnivorous wading birds? If yes, we developed field-based no effects and effects measures. This approach is analogous to the approach used to develop sediment-quality guidelines for the protection of aquatic life (see Long *et al*. 1995; MacDonald *et al*. 1996; MacDonald *et al*. 2000). If not, we proceeded to 5.

5. We derived a range within which the threshold for the receptor group of interest was expected to occur. Because information on the sensitivity of the receptor of interest was lacking, it was difficult to derive a threshold that was neither biased high or low. If bioassay data were available for several other species, however, one could calculate a threshold for each to determine a threshold range that spanned sensitive and tolerant species. That range was assumed to include the threshold for the receptor group of interest.

2.2.2.1 Mercury

Methylmercury is a strong nervous system toxicant. Its ability to cross the blood brain barrier results in brain lesions, damage to the central nervous system, and spinal cord degeneration (Wolfe *et al*. 1998). Methylmercury is absorbed into the bloodstream and transported to tissues and organs throughout the body (USEPA 1997b). As a result, neurological disorders, damage to organs, and effects on growth and development are characteristic effects of MeHg. Clinical symptoms of acute poisoning include ataxia, tremors, weakness in legs and wings, muscular incoordination, paralysis, recumbency, and convulsions (USEPA 1997b; Wolfe *et al*.

1998; Eisler 2001). Adverse effects also occur from chronic exposure to low concentrations of MeHg.

The reliance of piscivorus birds on fish makes them particularly susceptible to the adverse effects of MeHg toxicity. The proportion of mercury as MeHg in fish tissues is generally greater than 90%; and increases with fish length, weight, and age (Eisler 2001). Concentration data are in wet weight (ww), unless noted otherwise.

Survival

Studies by Spalding *et al*. (2000a; 2000b) and Bouton *et al.* (1999) found behavioral abnormalities and neurologic disturbances in great egrets (*Ardea albus*) dosed with 0.5 or 5 mg MeHgCl/kg. Birds had dingy feathers, avoided sun, and were less motivated to hunt. Great egrets in the high dose group experienced severe ataxia, as well as hematologic and histologic changes. Scheuhammer (1988) observed signs of mercury poisoning in Zebra finches (*Poephila guttata*) fed 5 mg MeHg/kg dry weight (dw). On day 40 of the 77 d study, some finches began exhibiting behavioral abnormalities. Symptoms included lethargy, fluffed feathers, and difficulty flying. The first death occurred on day 68 and by day 77 four of eight finches in the dose group had died with the rest showing neurological signs of mercury poisoning (Scheuhammer 1988). Neurological signs of poisoning did not appear until mercury concentrations were $\geq$15 mg/kg in the brain and between 30 and 40 mg/kg in the liver and kidneys. The high metabolism of the Zebra finch forces it to consume a greater amount of food, and as a result, mercury. This characteristic is similar to the belted kingfisher, which has a high food intake rate on a body weight basis (USEPA 1997b).

Neurological effects and death resulted from dietary treatments of 7.2 and 10 mg/kg of MeHg dicyandiamide fed to red-tailed hawks (*Buteo jamaicensis*; Fimreite and Karstad 1971). Birds that died showed symptoms similar to those described earlier:

muscular weakness, in-coordination, weight loss. Liver mercury residues ranged from 17 to 20 mg/kg in chicks that died. Lesions of axons and myelin sheaths were found in all hawks fed 7.2 and 10 mg/kg (Fimreite and Karstad 1971). Hill and Soares (1984) found similar effects in coturnix (*Coturnix japonica*) at a diet of 8 mg MeHgCl/kg over a 9 week period. During week 8, one male and three female birds began losing muscular control. One female died during week 9 after displaying signs of severe mercury toxicity. No clinical signs were shown over the 9 wk study in coturnix fed diets containing 0.125 or 4 mg MeHgCl/kg (Hill and Soares 1984). Hill and Soares (1984) established a single oral dose LD_{50} of 18 mg/kg body weight (bw) and an LC_{50} of 47 mg/kg.

Borg *et al*. (1970) fed juvenile goshawks (*Accipiter g. gentilis l.*) liver and muscle from MeHg contaminated chickens. The MeHg concentration in chicken muscle was 10 mg/kg and in liver 40 mg/kg. Diets contained either contaminated muscle and liver for a concentration of 13 mg/kg or muscle only for an average concentration of 10 mg/kg. The estimated intake of MeHg by goshawks was 0.7-1.2 mg/kg/day. Symptoms of MeHg poisoning were observed after a 2 week latency period and included inappetence, muscular weakness, ataxia, and loss of body weight. Goshawks in the 13 mg/kg group died at 30, 38, and 47 d. One bird in the 10 mg/kg group died on day 39. MeHg concentrations in the brains of these birds ranged from 26 to 46 mg/kg and in the livers from 96 to 138 mg/kg (Borg *et al*. 1970). Concentrations similar to those used by Borg *et al*. (1970) also proved fatal to pheasants (Spann *et al*. 1972). In their study, pheasants fed 30 mg/kg ethyl mercury p-toluene, equivalent to 12.5 mg Hg/kg, died between 57 and 102 days of feeding (Spann *et al.*1972). Symptoms leading up to death were similar to previously mentioned studies.

Reproduction

Mercury's potent embryo toxicity makes reproduction one of the most sensitive endpoints. Mercury concentrations well below those required to cause effects in adults can negatively impact reproduction and survival of young (Scheuhammer 1988; USEPA 1997b). Adverse effects of mercury on reproduction include reduced hatchability caused by increased mortality of embryos, smaller clutch sizes, a greater number of eggs laid outside the nest, and aberrant behavior (USEPA 1997b; Wolfe *et al.* 1998; Eisler 2001). Reproductive effects also extend to juvenile survival (Wolfe *et al*. 1998).

Adult mallard ducks (*Anas platyrhynchos*) were unaffected by mercury concentrations of 0.5 and 3 mg MeHg/kg dw, however, effects on reproduction were evident (Heinz 1979). Dosed hens laid more eggs outside the nestbox than controls, laid fewer sound eggs, and had less ducklings survive past one week (Heinz 1979). Duckling behavior was also affected. Treated ducklings had longer response times to tape recorded maternal calls than controls (Heinz 1979). Reproduction was similarly effected in black ducks (*Anas rubripes*) fed 3 mg MeHg dicyandiamide/kg over two reproductive seasons (Finley and Stendell 1978). The most harmful effects were on hatchability and duckling survival. Other effects were observed for clutch size, egg production, and the number of eggs incubated. Mercury concentrations in whole embryos that failed to hatch averaged 9.62 and 6.08 mg/kg for the first and second year, respectively. Brain mercury concentrations in dead ducklings ranged from 3.25 to 6.98 mg/kg and displayed lesions associated with mercury poisoning (Finley and Stendell 1978). Fimreite (1971) found comparable effects in pheasants fed 2-3 mg MeHg dicyandiamide/kg for 12 weeks. The number of shell-less eggs increased and egg weights decreased, as did hatchability and the number of fertilized eggs. Mercury concentrations in unhatched eggs ranged from 0.5 to 1.5 mg/kg (Fimreite 1971). A dietary dose of 10 mg/kg of ethyl mercury *p*-toluene (mercury equivalent of 4.2

mg/kg) reduced egg production 50-80% and increased mortality in eggs that were laid (Spann *et al.* 1972). A sample of treated eggs had an average mercury concentration of 1.5 mg/kg, and a range of 0.3 to 3.1 mg/kg (Spann *et al*. 1972).

Field Surveys

Common loons (*Gavia immer*) in northwestern Ontario displayed reduced nest site fidelity and laid fewer eggs in areas where mercury concentrations in prey averaged >0.4 mg/kg. Adult loon brain concentrations of mercury between 2 and 3 mg/kg were also associated with adverse effects on reproductive behavior (Barr 1996). Monteiro and Furness (2001) observed no clinical signs of poisoning in a single oral dose experiment with MeHg on free-living Cory's Shearwater chicks (*Calonectris dimoedea*). Exposure levels ranged from 0.9 to 2.5 mg/kg body weight and the researchers noted that they were similar to the no adverse effects level (NOAEL) of 2.5 mg/kg found by Scheuhammer (1988). Monteiro and Furness (2001) estimated that the highest average mercury brain concentration in the experiment was 3.4 mg/kg. This concentration was based on a blood:brain ratio of 0.78 calculated from adult Cory's shearwaters concentrations.

Effects Metrics

The most sensitive responses in birds exposed to methylmercury are associated with reproduction following long-term exposures. None of the available reproduction studies included species that could be considered piscivorus birds or reasonable surrogates. Mallards are often used in laboratory studies, however, their foraging behaviour is considerably different from the hypothetical piscivorus receptor. The hypothetical piscivorus receptor bears much more of a resemblance to birds such as brown pelicans, osprey, belted kingfishers and terns. Its diet primarily consists of fish and lesser amounts of invertebrates. The lack of toxicity data for piscivorus birds exposed to methylmercury precludes the development of a dose-response curve (using either a single or multiple studies) or the derivation of a NOAEL and LOAEL for piscivorus birds (*i.e.*, Options 1-3 in the hierarchy of decision criteria for choosing effects metrics are unavailable).

No field data were available to develop field-based benchmarks for piscivorus birds, which eliminates Option 4 for choosing effects metrics.

The final option for choosing an effects metric for piscivorus birds exposed to methylmercury is to derive a range within which the threshold for this receptor group is expected to occur. The most sensitive reproductive response was observed in mallard ducks exposed to methylmercury for three generations (Heinz 1974; 1979; Heinz and Locke 1975). In this study, 0.5 mg Hg/kg (as methylmercury dicyanamide) led to small, but significant reductions in clutch size and duckling survival. Similarly, Fimreite (1971) estimated the threshold egg concentration for hatchability to be between 0.5 and 1.5 mg Hg/kg for ring-necked pheasants. It would therefore seem reasonable to select 0.5 mg Hg/kg in the diet as the lower bound of the threshold range for piscivorus birds exposed to methylmercury. This dietary concentration was multiplied by the food intake rate of mallard ducks (0.128 kg/day, as measured by Heinz 1979) and normalized to their body weight (1 kg, Heinz *et al*. 1989) to derive the corresponding dose:

$$LT = \left(\frac{0.500\, mg\ Hg}{kg\ diet} \times \frac{128\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \ 1.00\ kg\ BW$$
$$= 0.0640\ mg\ /\ kg\, bw\ /\ day$$

EQUATION #3

where *LT* is the lower threshold dose and *BW* is body weight.

Survival and reproductive data on effects reveal a broad range of bird taxa that are severely affected by dietary concentrations of methylmercury /10 mg/kg. None of the tested species, which included mallards (Heinz and Hoffman 1998), goshawks (Borg *et al*. 1970), ring-necked pheasants (Spann *et al*. 1972), white leghorn chickens (Scott 1977) and Japanese quail (Hill and Soares 1984; Scott 1977), were able to tolerate dietary concentrations of methylmercury close to or greater than 10 mg/kg. The highest level of methylmercury in the diet that did not cause adverse impacts to a test species was 6 mg/kg. The test species was red-tailed hawks (Fimreite and Karstad 1971). It would therefore seem reasonable to select 6 mg Hg/kg in the diet as representing the tolerant end of the threshold range for piscivorus birds exposed to methylmercury. This dietary concentration was multiplied by the food intake rate of red-tailed hawks (0.109 kg/day, Craighead and Craighead 1956) and normalized to their body weight (1.126 kg, Dunning 1984) to derive the corresponding dose:

$$UT = \left(\frac{6.00\, mg\ Hg}{kg\ diet} \times \frac{109\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \, 1.13\ kg\ BW$$
$$= 0.581\, mg\ /\ kg\, bw\ /\ day$$

EQUATION #4

where *UT* is the upper threshold dose and *BW* is body weight.

Therefore, the threshold range for piscivorus birds exposed to methylmercury is 0.0640 to 0.581 mg Hg/kg bw/day.

2.2.2.2 TCDD-TEQs

This section will examine the effects of TCDD (2,3,7,8-tetrachlorodibenzo-*p*-dioxin) and equivalents to piscivorus birds. The TCDD Equivalent (TEQ) approach relates the toxicity of specific PCB (polychlorinated biphenyl), PCDD (polychlorinated dibenzo-*p*-dioxin), and PCDF (polychlorinated dibenzofuran) congeners to that of TCDD. This technique provides a basis with which to compare the results of toxicity studies involving PCB, PCDD, and PCDF mixtures and congeners to the specific congener profiles of sites in the Calcasieu Estuary system and is described further in Appendix G. Literature relating to survival, growth, and reproduction was reviewed. The focal species in this section are belted kingfishers, osprey, terns, and brown pelicans. Additional species will be included in the discussion when necessary. Concentration data are in wet weight (ww), unless noted otherwise.

Survival

Nosek *et al*. (1992a) treated mature hen pheasants with single intraperitoneal TCDD injections of 6,250, 25,000, or 100,000 ng/kg bw. These birds suffered body weight loss and mortality at the two higher dose levels. All birds given the 100,000 ng/kg bw dose were dead by the sixth week and 75% of those given 25,000 ng/kg bw were dead by the twelfth week. These investigators also examined the subchronic effects of TCDD to pheasants, dosing birds weekly with 10, 100, or 1,000 ng/kg bw for ten weeks (cumulative doses of 100, 1,000, or 10,000 ng/kg bw). Fifty-seven percent of birds given the highest dose died within the 24 week experiment, while those on the lower doses experienced no mortality. Bobwhite quail, mallards and ringed turtledoves given single oral doses of TCDD were found to have 37-day LD_{50}s of 15,000, 108,000, and 810,000 ng/kg bw, respectively (Hudson *et al*. 1984). Chickens given single oral doses of 25,000 ng/kg died within 12 days (Grieg *et al*. 1973) and a 21 day oral NOAEL of 100 ng/kg/day was reported for treatments to 3 day old white leghorn chicks (Schwetz *et al*. 1973).

Reproduction

Nosek *et al*. (1992a) monitored egg production and embryonic mortality in mature hen ring-necked pheasants after weekly intraperitoneal injections of 10, 100, and 1,000 ng/kg bw TCDD. At the highest dose, egg production fell significantly compared to controls - from a cumulative total of 33 eggs per bird down to 12. Egg production was not affected at the two lower doses. Embryotoxicity significantly increased in response to the dose level. Cumulative doses of 100 and 1,000 ng/kg bw elicited insignificant increases in embryo mortality, but the 10,000 ng/kg bw dose caused 100% embryo mortality compared to 0% in controls.

The effects of egg injection of TCDD , TCDF, and PCBs have been reported by several investigators and include decreased egg production and increased embryonic mortality. Studies of egg injections with PCBs have demonstrated that when similar toxicant levels are attained in the egg via injection and via conventional maternal dietary doses, the effects to the chicks are also similar (Hoffman *et al*. 1996a; Nosek *et al*. 1993). Embryonic uptake of organochlorines from yolk is similar for substances injected into the yolk and for those accumulated naturally (Peakall and Fox 1987). Bioaccumulative environmental substances concentrate in egg yolks (Tumasonis *et al*. 1973; Custer *et al*. 1997). As a result, many studies have been conducted examining the effects of injecting environmentally relevant concentrations of PCBs into yolks. Egg yolk injected PCBs are distributed throughout the embryo, including fat tissue, liver, kidneys and bone marrow (Brunstrom *et al*. 1982). Ring-necked pheasant hens fed radiolabeled TCDD were found to eliminate approximately 1% of their body burdens into eggs, and all of the substance was deposited in the yolk, none in the albumin (Nosek *et al*. 1992b). The maternal transfer of total PCBs to eggs for several avian species was investigated by Drouillard and Norstrom (2001). Ratios of egg yolk to maternal adipose tissue PCB concentrations ranged from 0.270 in ring doves to 1.20 in chickens and pheasant.

Henshel *et al*. (1997) estimated the LD_{50} of TCDD injected into white leghorn chicken eggs yolks to be 122 ng/kg egg (by probit analysis, 146 ng/kg egg when determined by interpolation) and Powell *et al*. (1996b) observed that hatchability of white leghorn chicken eggs significantly decreased at a dose of 160 ng/kg egg TCDD injected into egg yolks. McKinney *et al*. (1976) reported that the injection of 5,000 ng/kg ww egg 2,3,7,8-TCDF (500 ng/kg ww egg TEQ) resulted in complete mortality of one-day-old white leghorn chickens within an average of 11.5 days. Chickens fed diets containing fish from a TCDD and PCB contaminated site at increasing concentrations experienced time and dose related decreases in egg hatchability (Summer *et al*. 1996). Total PCB concentrations in the diet ranged from 0.300 to 6.60 mg/kg. This corresponds to concentrations of 3.3 to 59 ng/kg diet of TCDD, determined by the H4IIE bioassay.

Cormorant eggs were less sensitive to the effects of TCDD. Eggs collected from an isolated colony in Manitoba were injected with 4,000 ng/kg TCDD into the yolk sac. Cormorant eggs receiving 4,000 ng/kg egg TCDD suffered 50% mortality while controls experienced 28% mortality (Powell *et al*. 1997) in one experiment. The investigators then increased the dose range in a subsequent study (Powell *et al*. 1998) and observed 44.7% mortality in controls and 84.9% in eggs treated with 11,900 ng/kg egg TCDD. The LD_{50} of this second study was estimated to be 4,000 ng/kg egg.

Nosek *et al*. (1993) estimated a TCDD LD_{50} of 2,180 ng/kg ww egg to ring-necked pheasants when administered in egg yolks. Eggs were injected on day 0 of embryonic development with doses of 10, 100, 1,000, or 10,000 ng/kg egg and mortality, defined as a "failure of the hatchling to emerge completely from the shell alive", was monitored. Eggs treated with the three lowest doses showed no significant increase in embryo mortality over controls, while the 10,000 ng/kg dose caused near total (98%) embryonic failure.

Chicken eggs that had been incubating for four days were injected with 3,3',4,4',5-pentachlorobiphenyl (PCB126) at treatment levels ranging from 0 to 2,000 ng/kg (0 to 200 ng/kg WHO TEQ; Brunstrom and Anderson 1988). After 14 days, embryonic mortality was highest in the highest treatment group (90%) compared to control groups (vehicle only = 15% embryo mortality). Brunstrom (1989) found that PCB126 was the most toxic of the congeners tested, with PCB77, 105, and 118 being 5, 1,000, and 8,000 times less toxic, respectively.

Powell *et al*. (1996a; 1996b; 1997) also investigated the embryotoxicity of PCB126 to chickens, and cormorants. Chicken eggs were yolk-injected with 100,000, 200,000, 400,000, 800,000, 1,600,000, 3,200,000, 6,400,000, and 12,800,000 ng/kg egg prior to incubation. The LD_{50} for chick embryos was estimated to be 2,300 ng/kg egg (230 ng/kg egg TEQ). Cormorant eggs were collected from Lake Winnipegosis in Manitoba, Canada and were injected with doses of PCB126 at levels of 0, 5,000, 10,000, 25,000, 50,000, 100,000, 200,000, 400,000, and 880,000 ng/kg ww egg (0 to 88,000 ng/kg ww egg TEQ). The eggs were then incubated for 21 days and candled on days 7, 14, and 21 to check for viability. Significant increases in embryo mortality were observed in the 400,000 and 880,000 ng/kg dose groups (40,000 and 88,000 ng/kg TEQ), to 87% and 100%, respectively (Powell *et al*. 1997). An LD_{50} of 158,000 ng/kg (15,800 ng/kg TEQ) was estimated. A second study involving cormorants estimated a PCB126 LD_{50} of 177,000 ng/kg ww egg (17,700 ng/kg ww egg TEQ) after a single injection into the yolk (Powell *et al*. 1998).

PCB77 is another PCB congener whose toxicity closely resembles that of TCDD. Chicken eggs injected into the yolk with 5,000 or 20,000 ng/kg egg (250 or 1,000 ng/kg egg TEQ) showed significantly higher embryonic mortality (55 and 100%) than in controls (15%). Herring gull and goose eggs injected with doses as high as 1,000,000 ng/kg egg (50,000 ng/kg egg TEQ) showed no significant increases in mortality and duck eggs showed no significant increases in mortality with doses as high as 5,000,000 ng/kg egg (250,000 ng/kg egg TEQ; Brunstrom 1988). Wild turkey embryos were also much less sensitive to PCB77 than were chickens. The Ah receptor, thought to be instrumental in the expression of TCDD and PCB toxicity, is not present in turkeys in the embryonic stage of development, and may therefore provide a basis for the species difference (Brunstrom and Lund 1988).

Henshel *et al.*(1997) compared the relative sensitivities of TCDD yolk and air sac injections into the eggs of white leghorn chickens. Eggs were injected on day 0 of embryonic development and were allowed to hatch undisturbed. The result was a significantly (60%) lower LD_{50} for the yolk route of administration. Air sac injections of PCB126 were also investigated by Hoffman *et al*. (1998) in multiple bird species. White leghorn chicken embryo was the most sensitive with an LD_{50} of 400 ng/kg ww egg (40 ng/kg ww egg TEQ), while American kestrel and common tern embryos were less sensitive with LD_{50}s of 65,000 and 104,000 ng/kg ww egg (6,500 and 10,400 ng/kg ww egg TEQ), respectively. An LD_{50} of 8,600 ng/kg egg (430 ng/kg TEQ) was calculated for chick eggs dosed with PCB77 administered into the air sac (Brunstrom and Anderson 1988).

Other Effects

Henshel (1998) dosed white leghorn chicken embryos with TCDD via yolk injection and examined the symmetry of the tectum and forebrain of the chicks' brains. Chickens suffered brain deformities, as asymmetries, at doses as low as 10 ng TCDD/kg egg administered via egg yolk injection. Herons and cormorants showed brain asymmetry at accumulated TCDD levels of 10 and 19 ng/kg egg. Investigations of the teratogenic effects of PCB126 in chicks revealed the potential for beak deformities and edema (Powell *et al*. 1996a; 1996b). Injections of PCB126 at levels of 900 ng/kg egg (90 ng/kg egg TEQ) caused a significant increase in the number of abnormal embryos per number of eggs (13/60 vs 3/59 for the vehicle control) while having no significant impact on mortality of the birds. Other abnormalities noted included small or missing eyes and curved toes.

Weight gain of chicks is also an effect of PCB and TCDD exposure. White leghorn cockerels were fed a variety of hexachlorobiphenyl congeners at 400,000,000 ng/kg diet for 21 days and body weights monitored (McKinney *et al*. 1976). Three of five congeners tested significantly inhibited weight gain of the birds, with 2,4,5,2',4',5'-HCB having the most impact (chick weight 78% of controls on day 21). One congener, 3,4,5,3',4',5'-HCB, produced 100% mortality in test animals within 11 days of the onset of the experiment. Nestling kestrels orally dosed with PCB126 to levels of 50,000, 250,000, and 1,000,000 ng/kg bw/day (5,000, 25,000 and 100,000 ng/kg bw/day TEQ) via the diet also experienced inhibited weight gain (Hoffman *et al*. 1996b). For days 4 to 10 of the study, there was a significant correlation between PCB concentration and decreased body weight. Smaller bone lengths also indicated a reduced growth rate. Humerus, radius-ulna, and tibiotarsus were all significantly shorter in the 250,000 and 1,000,000 ng/kg bw/day (25,000 and 100,000 ng/kg bw/day TEQ) test groups than controls. Embryonic exposure to PCB126 also resulted in decreased growth rates in white leghorn chickens (Powell *et al*. 1996a). Injection of 900 ng/kg egg (90 ng/kg egg TEQ) of PCB126 prior to incubation produced significantly reduced body weights by the second week and 3,000 ng/kg egg (150 ng/kg egg TEQ) of PCB77 reduced body weights compared to controls at 3 weeks (Powell *et al*. 1996a).

Field Studies

There has been some discussion in the literature regarding the relationship between adverse reproductive effects to birds observed in the field and long-lived chlorinated organic pollutants (de Voogt *et al*. 2001). PCBs and DDE are ubiquitous pollutants found at many contaminated sites. Cormorants in the Great Lakes area have a strong correlation (r^2 = 0.703) between egg mortality and bioassay-derived dioxin equivalents (Tillitt *et al*. 1992). Custer *et al*. (1999) instead suggest that DDE was primarily responsible for the observations of *in situ* decline in cormorant reproductive success in this area and that TCDD equivalents did not have a significant effect on cormorant reproductive success in Green Bay, despite significant PCB contamination. Eggs containing 299 ng TEQ/kg egg had 39% mortality while eggs from the reference site, containing only 35 ng TEQ/kg egg had 8% mortality. DDE concentrations were not included in the analyses.

Elliott *et al*. (2001) investigated the effects of organochlorine substances on the reproductive success of osprey in the Fraser and Columbia river systems. Analysis of concentrations in egg yolks and the results of laboratory incubation of eggs from the test and reference sites showed no correlation between embryonic mortality and *in ovo* substance exposure, despite hatching success ranging from 56 to 100% at the various sites. Woodford *et al*. (1998) monitored reproductive success of osprey exposed to chlorinated substances in the Wisconsin River from 1992 to 1996. Study sites included two test sites downstream of two bleached-kraft facilities and two reference sites upstream. From these sites, eggs were collected to measure contamination levels and the remaining eggs monitored for hatching and fledging rates as well as weight gain. Exposure to PCDDs, PCDFs and coplanar PCBs at these sites did not affect hatching or fledging rates, but chick growth may have been reduced at TCDD concentrations ranging from 54 to 67 ng/kg ww egg.

PCBs and dioxins have been linked to teratogenic effects in the field. Ludwig *et al*. (1996) observed a relationship between the abnormality rate (number of abnormalities per 1000 eggs) and TEQ concentrations in double-crested cormorants and Caspian terns in the upper Great Lakes. Bill defects and edema were the most common deformities. The abnormality rate reached 14.3% in live cormorant eggs in Green Bay, WI and 28.8% in live tern eggs in Saginaw Bay, MI, both contaminated sites. The overall live cormorant egg deformity rate correlated positively with TEQs (r^2 = 0.86) and the overall live tern egg deformity rate did not correlate as well with TEQs (r^2 = 0.12).

Effects Metrics

The most sensitive responses in birds exposed to TCDD-TEQs are associated with reproduction following long-term exposure. None of the available reproduction studies included species that could be considered piscivorus birds or reasonable surrogates. The hypothetical receptor embodies characteristics similar to brown pelicans, osprey, belted kingfishers, and terns. The lack of toxicity data for piscivorus birds exposed to TCDD-TEQs precludes the development of a dose-response curve (using either a single or multiple studies) or the derivation of a NOAEL and LOAEL for piscivorus birds (i.e., Options 1-3 in the hierarchy of decision criteria for choosing effects metrics are unavailable). The field data were insufficient to develop field-based benchmarks for piscivorus birds of the Calcasieu Estuary, which eliminates Option 4 for choosing effects metrics.

The final option for choosing an effects metric for piscivorus birds exposed to TCDD-TEQs is to derive a range within which the threshold for this receptor group is expected to occur. The most sensitive response observed was reproductive success of ring-necked pheasants injected weekly with TCDD (Nosek *et al*. 1992a). In this study, 14 ng TCDD/kg bw/day did not significantly reduce reproductive success of hen pheasants, while the next highest dose of 140 ng TCDD/kg bw/day caused a decrease in cumulative egg production. This concentration will be used to represent the sensitive end of the toxicity threshold.

The upper bound of the threshold range is derived from a study on the effects of PCB126 to American kestrel hatchlings (Hoffman 1996a). In this study, the highest level of TEQ in the diet that did not cause adverse effects was 5,000 ng/kg TEQ. This dietary concentration is multiplied by the food intake rate of American kestrel hatchlings (0.00778 kg/day, Nagy 1987) and normalized to their body weight (0.076 kg, Hoffman *et al*. 1996b) to derive the corresponding dose:

$$UT = \left(\frac{5{,}000\ ng\ TEQ}{kg\ diet} \times \frac{7.78\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \ 0.0760\ kg\ BW$$
$$= 512\ ng / kg bw / day$$

EQUATION #5

where *UT* is the upper threshold dose and *BW* is body weight.

Therefore, the threshold range for sediment-probing birds exposed to TCDD-TEQs is 14 to 512 ng TEQ/kg bw/day.

2.2.3 Risk Characterization

In the risk characterization phase of the probabilistic risk assessment, the results of the exposure assessment (i.e., reverse cumulative distribution functions) and effects measures are integrated to develop risk estimates.

3.0 Results and Discussion

3.1 Probabilistic Ecological Risk Assessment

Mercury – Bayou d'Inde AOC

The Monte Carlo analysis revealed that exposure of average-sized carnivorous wading birds to mercury in BI AOC could range from a minimum of 0.0029 to a maximum of 0.0747 mg/kg bw/day. The mean exposure is 0.0166 mg/kg bw/day and the median exposure is 0.0152 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0075 and 0.0308 mg/kg bw/day. Figure H2-4 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by the slope term of FMR (r_p = 0.54), and gross energy of fish (r_p = -0.44).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-4. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000978 and 0.0566 mg/kg bw/day. The 50th percentile ranges between 0.00182 and 0.0956 mg/kg bw/day, and the 90th percentile ranges between 0.00303 and 0.187 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.0088, the 50th percentile is 0.0153, and the 90th percentile is 0.0268 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small carnivorous wading birds to mercury could range from a minimum of 0.0032 to a maximum of 0.0872 mg/kg bw/day. The mean exposure is 0.0188 mg/kg bw/day and the median exposure is 0.0173 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0086 and 0.034 mg/kg bw/day. Figure H2-5 depicts the cumulative distribution of mercury intake rates for the hypothetical small carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by the slope term of FMR (r_p = 0.56), and gross energy of fish (r_p = -0.46).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-5. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.00114 and 0.065 mg/kg bw/day. The 50th percentile ranges between 0.00208 and 0.108 mg/kg bw/day, and the 90th percentile ranges between 0.00343 and 0.209 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.011, the 50th percentile is 0.0184, and the 90th percentile is 0.0293 mg/kg bw/day.

Mercury – Middle Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized carnivorous wading birds to mercury could range from a minimum of 0.0010 to a maximum of 0.0317 mg/kg bw/day. The mean exposure is 0.0065 mg/kg bw/day and the median exposure is 0.0060 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0030 and 0.0118 mg/kg bw/day. Figure H2-6 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by the slope term of FMR (r_p = 0.54), and gross energy of fish (r_p = -0.44).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-6. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000315 and 0.0196 mg/kg bw/day. The 50th percentile ranges between 0.000594 and 0.033 mg/kg bw/day, and the 90th percentile ranges between 0.00099 and 0.0653 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00343, the 50th percentile is 0.00597, and the 90th percentile is 0.0102 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small carnivorous wading birds to mercury could range from a minimum of 0.0014 to a maximum of 0.0332 mg/kg bw/day. The mean exposure is 0.0074 mg/kg bw/day and the median exposure is 0.0068 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0034 and 0.0133 mg/kg bw/day. Figure H2-7 depicts the cumulative distribution of mercury intake rates for the hypothetical small carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.58], followed by the slope term of FMR (r_p = 0.55), and gross energy of fish (r_p = -0.46).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-7. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000367 and 0.0225 mg/kg bw/day. The

50th percentile ranges between 0.000681 and 0.0373 mg/kg bw/day, and the 90th percentile ranges between 0.00111 and 0.073 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.004, the 50th percentile is 0.00674, and the 90th percentile is 0.0115 mg/kg bw/day.

Mercury – Upper Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized carnivorous wading birds to mercury could range from a minimum of 0.0009 to a maximum of 0.0203 mg/kg bw/day. The mean exposure is 0.0052 mg/kg bw/day and the median exposure is 0.0048 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0024 and 0.0097 mg/kg bw/day. Figure H2-8 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.61], followed by the slope term of FMR (r_p = 0.53), and gross energy of fish (r_p = -0.44).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-8. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000278 and 0.0173 mg/kg bw/day. The 50th percentile ranges between 0.000524 and 0.0291 mg/kg bw/day, and the 90th percentile ranges between 0.000868 and 0.0578 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00275, the 50th percentile is 0.00475, and the 90th percentile is 0.00821 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small carnivorous wading birds to mercury could range from a minimum of 0.0012 to a maximum of 0.0262 mg/kg bw/day. The mean exposure is 0.0059 mg/kg bw/day and the median exposure is 0.0054 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0027 and 0.0107 mg/kg bw/day. Figure H2-9 depicts the cumulative distribution of mercury intake rates for the hypothetical small carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by the slope term of FMR (r_p = 0.54), and gross energy of fish (r_p = -0.46).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-9. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000323 and 0.02 mg/kg bw/day. The 50th percentile ranges between 0.0006 and 0.033 mg/kg bw/day, and the 90th percentile ranges between 0.000982 and 0.0646 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.0032, the 50th percentile is 0.00544, and the 90th percentile is 0.00943 mg/kg bw/day.

Mercury – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized carnivorous wading birds to mercury could range from a minimum of 0.0005 to a maximum of 0.0132 mg/kg bw/day. The mean exposure is 0.0029 mg/kg bw/day and the median exposure is 0.0027 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0013 and 0.0054 mg/kg bw/day. Figure H2-10 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by the slope term of FMR (r_p = 0.53), and gross energy of fish (r_p = -0.44).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-10. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.00014 and 0.00948 mg/kg bw/day. The 50th percentile ranges between 0.000268 and 0.0159 mg/kg bw/day, and the 90th percentile ranges between 0.000441 and 0.032 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00158, the 50th percentile is 0.00278, and the 90th percentile is 0.004656 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small carnivorous wading birds to mercury could range from a minimum of 0.0005 to a maximum of 0.0129 mg/kg bw/day. The mean exposure is 0.0033 mg/kg bw/day and the median exposure is 0.0030 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0015 and 0.006 mg/kg bw/day. Figure H2-11 depicts the cumulative distribution of mercury intake rates for the hypothetical small carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by the slope term of FMR (r_p = 0.54), and gross energy of fish (r_p = -0.45).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-11. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000163 and 0.0109 mg/kg bw/day. The

50th percentile ranges between 0.000307 and 0.0179 mg/kg bw/day, and the 90th percentile ranges between 0.000499 and 0.0358 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00179, the 50th percentile is 0.00304, and the 90th percentile is 0.00521 mg/kg bw/day.

TCDD-TEQs – Bayou d'Inde AOC

The Monte Carlo analysis revealed that exposure of average-sized carnivorous wading birds to TCDD-TEQs could range from a minimum of 1.29 to a maximum of 39.7 ng/kg bw/day. The mean exposure is 8.3 ng/kg bw/day and the median exposure is 7.59 ng/kg bw/day. Ninety percent of exposure estimates are between 3.67 and 15.2 ng/kg bw/day. Figure H2-12 depicts the cumulative distribution of TCDD-TEQs intake rates for the hypothetical average-sized carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.63], followed by the slope term of FMR (r_p = 0.54), and gross energy of fish (r_p = -0.44).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-12. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 1.63 and 165 ng/kg bw/day. The 50th percentile ranges between 2.9 and 281 ng/kg bw/day, and the 90th percentile ranges between 4.9 and 529 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 3.46, the 50th percentile is 5.59, and the 90th percentile is 9.1 ng/kg bw/day.

The Monte Carlo analysis revealed that exposure of small carnivorous wading birds to TCDD-TEQs could range from a minimum of 1.61 to a maximum of 38.7 ng/kg bw/day. The mean exposure is 9.39 ng/kg bw/day and the median exposure is 8.64 ng/kg bw/day. Ninety percent of exposure estimates are between 4.29 and 17 ng/kg bw/day. Figure H2-13 depicts the cumulative distribution of TCDD-TEQs intake rates for the hypothetical small carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by the slope term of FMR (r_p = 0.56), and gross energy of fish (r_p = -0.46).

The probability bounds estimated for small carnivorous wading birds are depicted in Figure H2-13. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 1.4 and 144 ng/kg bw/day. The 50th percentile ranges between 2.54 and 248 ng/kg bw/day, and the 90th percentile ranges between 4.33 and 473 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 4.29, the 50th percentile is 7.55, and the 90th percentile is 13 ng/kg bw/day.

TCDD-TEQs – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized carnivorous wading birds to TCDD-TEQs could range from a minimum of 0.31 to a maximum of 12.7 ng/kg bw/day. The mean exposure is 2.44 ng/kg bw/day and the median exposure is 2.24 ng/kg bw/day. Ninety percent of exposure estimates are between 1.11 and 4.51 ng/kg bw/day. Figure H2-14 depicts the cumulative distribution of TCDD-TEQs intake rates for the hypothetical average-sized carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) =

0.61], followed by the slope term of FMR (r_p = 0.53), and gross energy of fish (r_p = -0.46).

The probability bounds estimated for average-sized carnivorous wading birds are depicted in Figure H2-14. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.455 and 16.4 ng/kg bw/day. The 50th percentile ranges between 0.823 and 28.3 ng/kg bw/day, and the 90th percentile ranges between 1.41 and 53.9 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.68, the 50th percentile is 1.11, and the 90th percentile is 1.82 ng/kg bw/day.

The Monte Carlo analysis revealed that exposure of small carnivorous wading birds to TCDD-TEQs could range from a minimum of 0.561 to a maximum of 11.2 ng/kg bw/day. The mean exposure is 2.77 ng/kg bw/day and the median exposure is 2.55 ng/kg bw/day. Ninety percent of exposure estimates are between 1.27 and 5 ng/kg bw/day. Figure H2-15 depicts the cumulative distribution of TCDD-TEQs intake rates for the hypothetical small carnivorous wading bird species.

Sensitivity analysis revealed that the power term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by the slope term of FMR (r_p = 0.55), and gross energy of fish (r_p = -0.45).

The probability bounds estimated for small carnivorous wading birds are depicted in Figure H2-15. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.53 and 18.8 ng/kg bw/day. The 50th percentile ranges between 0.941 and 32 ng/kg bw/day, and the 90th percentile ranges between 1.59 and 60.3 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 1.27, the 50th percentile is 2.23, and the 90th percentile is 3.83 ng/kg bw/day.

3.2 Risk Assessment

For the AOCs and reference areas of the Calcasieu Estuary, a low, indeterminate, and high category of risk was determined for carnivorous wading birds for each COC. These categories of risk were derived using the following guidance:

1. If the probability of exceeding the lower toxicity threshold was less than 20%, the risk was considered low;
2. If the probability of exceeding the upper toxicity threshold was greater than 20%, then the risk was considered high; and,
3. All other probabilities we considered to have indeterminate risk.

Mercury – Bayou d'Inde AOC

The Monte Carlo predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in BI AOC. Therefore, mercury poses low risk to the survival and reproduction of average-sized carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be as much as 82% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for average-sized carnivorous wading birds exposed to mercury in BI AOC (Table H2-4).

The Monte Carlo analysis indicates that average-sized carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

The Monte Carlo predictions for total daily intake rates of mercury by small birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in BI AOC. Therefore, mercury poses low risk to the survival and reproduction of small carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be 91% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for small carnivorous wading birds exposed to mercury in BI AOC (Table H2-4).

The Monte Carlo analysis indicates that small carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

Mercury – Middle Calcasieu River AOC

The Monte Carlo predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in MCR AOC. Therefore, mercury poses low risk to the survival and reproduction of average-sized carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be as much as 11% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for average-sized carnivorous wading birds exposed to mercury in MCR AOC (Table H2-4).

The Monte Carlo analysis indicates that average-sized carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 81% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

The Monte Carlo predictions for total daily intake rates of mercury by small birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in Middle Calcasieu River. Therefore, mercury poses low risk to the survival and reproduction of small carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be 15% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for small carnivorous wading birds exposed to mercury in MCR AOC (Table H2-4).

The Monte Carlo analysis indicates that small carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 95% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

Mercury – Upper Calcasieu River AOC

The Monte Carlo predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in UCR AOC. Therefore, mercury poses low risk to the survival and reproduction of average-sized carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be 7% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for average-sized carnivorous wading birds exposed to mercury in UCR AOC (Table H2-4).

The Monte Carlo analysis indicates that average-sized carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 81% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

The Monte Carlo predictions for total daily intake rates of mercury by small birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in UCR AOC. Therefore, mercury poses low risk to the survival and reproduction of small carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be 10% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for small carnivorous wading birds exposed to mercury in UCR AOC (Table H2-4).

The Monte Carlo analysis indicates that small carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 89% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

Mercury – Reference Areas

The Monte Carlo predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in reference areas. Therefore, mercury poses low risk to the survival and reproduction of average-sized carnivorous wading birds. The probability bounds also indicated that there is no chance that the upper or lower toxicity threshold will be exceeded (Table H2-4).

The Monte Carlo analysis indicates that average-sized carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark (Table H2-5).

The Monte Carlo predictions for total daily intake rates of mercury by small birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in reference areas. Therefore, mercury poses low risk to the survival and reproduction of small carnivorous wading birds. The probability bounds also indicated that there is no chance that the upper or lower toxicity threshold will be exceeded (Table H2-4).

The Monte Carlo analysis indicates that small carnivorous wading birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark (Table H2-5).

TCDD-TEQs – Bayou d'Inde AOC

The Monte Carlo predictions for total daily intake rates of TCDD-TEQs by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in BI AOC. Therefore, TCDD-TEQs pose low risk to the survival and reproduction of average-sized carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be as much as 100% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 11%. Thus, there is some uncertainty regarding the low risk conclusion for average-sized carnivorous wading birds exposed to TCDD-TEQs in BI AOC (Table H2-4).

The Monte Carlo analysis indicates that average-sized carnivorous wading birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

The Monte Carlo predictions for total daily intake rates of TCDD-TEQs by small birds indicate that there is 7.4% probability that the lower toxicity thresholds is being exceeded in BI AOC. The probability of exceeding the upper threshold was 0%. Therefore, TCDD-TEQs pose low risk to the survival and reproduction of small carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be 100% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 8%. Thus, there is some uncertainty regarding the low risk conclusion for small carnivorous wading birds exposed to TCDD-TEQs in BI AOC (Table H2-4).

The Monte Carlo analysis indicates that small carnivorous wading birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

TCDD-TEQs – Reference Areas

The Monte Carlo predictions for total daily intake rates of TCDD-TEQs by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in reference areas. Therefore, TCDD-TEQs pose low risk to the survival and reproduction of average-sized carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be as much as 96% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for average-sized carnivorous wading birds exposed to TCDD-TEQs in reference areas (Table H2-4).

The Monte Carlo analysis indicates that average-sized carnivorous wading birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 18% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

The Monte Carlo predictions for total daily intake rates of TCDD-TEQs by small birds indicate that there is 0% probability that the lower toxicity thresholds is being exceeded in reference areas. The probability of exceeding the upper threshold was 0%. Therefore, TCDD-TEQs pose low risk to the survival and reproduction of small carnivorous wading birds. However, the upper probability bound suggests (given our incertitude) that there could be 99% probability that the lower toxicity threshold is being exceeded. The probability of exceeding the upper threshold was 0%. Thus, there is some uncertainty regarding the low risk conclusion for small carnivorous wading birds exposed to TCDD-TEQs in reference areas (Table H2-4).

The Monte Carlo analysis indicates that small carnivorous wading birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 24% probability of exceeding the Appendix G benchmark, respectively (Table H2-5).

Review of historical data associated with CH2M Hill's Calcasieu Estuary Biological Monitoring Program indicated no invertebrate data for COCs that screened through for this assessment. Most of the reported data were for fish, which are not a part of the diet of sediment-probing birds.

4.0 Uncertainty Analysis

There are a number of sources of uncertainty in the assessment of risk of COCs to carnivorous wading birds, including uncertainties in the conceptual model, in the exposure assessment, and in the effects assessment. As each of these sources of uncertainty can influence the estimations of risk, it is important to describe and, when possible, quantify the magnitude and direction of such uncertainties. In this way, it is possible to evaluate the level of confidence that can be placed in the assessments conducted using the various lines of evidence.

Uncertainties Associated with the Conceptual Model - The conceptual model is intended to define the linkages between stressors, potential exposure, and predicted effects on ecological receptors. As such, the conceptual model provides the scientific basis for selecting assessment and measurement endpoints to support the risk assessment process. Potential uncertainties arise from lack of knowledge regarding ecosystem functions, failure to adequately address spatial and temporal variability in the evaluations of sources, fate, and effects, omission of stressors, and overlooking secondary effects. The types of uncertainties associated with the conceptual model that links contaminant sources to effects on carnivorous wading birds include those associated with the identification of COCs, environmental fate and transport of COCs, exposure pathways, receptors at risk, and ecological effects. Of these, the identification of exposure pathways probably represents the primary source of uncertainty in the conceptual model. In this assessment, it was assumed that exposure to contaminated food represents the most important pathway for exposing carnivorous wading birds to COCs. Other pathways of exposure are likely minor, but could increase risks somewhat.

Uncertainties Associated with the Exposure Assessment - The exposure assessment is intended to describe the actual or potential co-occurrence of stressors with receptors. As such, the exposure assessment identifies the exposure pathways and the intensity and extent of contact with stressors for each receptor or group of receptors at risk. There are a number of potential sources of uncertainty in the exposure assessment, including measurement errors, extrapolation errors, and data gaps.

In this assessment, one type of measurement was used to evaluate exposure of carnivorous wading birds to COCs and this includes the chemical analyses of tissue residues in fish. Analytical errors and descriptive errors represent potential sources of uncertainty.

Three approaches were used to address concerns relative to these sources of uncertainty.

First, analytical errors were evaluated using information on the accuracy, precision, and detection limits (DL) generated to support the Phase I and Phase II sampling programs. The results of this analysis indicated that most of the data used in this assessment met the project data quality objectives. Second, all data entry, data translation, and data manipulations were audited to ensure their accuracy. Data auditing involved 10% number-for-number checks against the primary data source initially, increasing to 100% number-for-number checks if significant errors were detected in the initial auditing step. Finally, statistical analyses of data were conducted to evaluate data distributions, identify the appropriate summary statistics to generate, and evaluate the variability in the observations. As such measurement errors in the tissue residue data are considered to be of minor important and are unlikely to influence the results of the risk assessment. Data gaps represent a source of uncertainty in the assessments of exposure for carnivorous wading birds.

Uncertainties in the Effects Assessment **-** The effects assessment is intended to describe the effects caused by stressors, link them to the assessment endpoints, and evaluate how effects change with fluctuations in the levels (i.e., concentrations) of the various stressors. There are several sources of uncertainty in the assessment of effects including measurement errors, extrapolation errors, and data gaps.

No effects data were available for carnivorous wading birds exposed to COCs. As a result, threshold ranges were developed using laboratory effects data on other bird species. This approach introduces uncertainty due to extrapolation between species and extrapolation from laboratory conditions in the Calcasieu Estuary.

5.0 Conclusions

Mercury

The risk characterization results indicate that there is little chance of mercury exposure exceeding the effects thresholds for carnivorous wading birds in the Calcasieu Estuary AOCs. Thus, carnivorous wading birds are at a low risk of adverse effects associated with exposure to mercury in the Calcasieu Estuary AOCs and reference areas. There is, however, some uncertainty regarding this conclusion in BI AOC.

TCDD-TEQs

There is no chance of TCDD-TEQ exposure exceeding the lower or upper effects thresholds for carnivorous wading birds foraging in Calcasieu Estuary AOCs. Therefore, TCDD-TEQs pose low risks to the survival and reproduction of carnivorous wading birds in the Calcasieu Estuary AOCs and reference areas. There is, however, considerable uncertainty regarding this conclusion in both, BI AOC and the reference areas.

Probabilistic Risk Assessment Limitations

There are several limitations of the probabilistic risk analyses that influence our confidence regarding the above risk statements. These include:

- The sensitivity analyses for the Monte Carlo simulations indicated that the most important input variables were the slope and power terms used to estimate free metabolic rate (*FMR*). The *FMR* used in the analyses was based on the allometric equation from Nagy (1987). No corresponding measurements of this variable are available for carnivorous wading bird species. The potential magnitude and direction of the uncertainty associated with lack of empirical data on metabolic rate are unknown. We did, however, investigate the possible consequences of the uncertainty in this variable due to model error (i.e., the error associated with the lack of fit of the allometric model that relates *FMR* to body weight). This source of uncertainty did not strongly impact our conclusions regarding risk.

- Sample size for COCs in fish and invertebrate tissues was generally limited. Although we accounted for this source of uncertainty in our analyses, it is possible that additional data would substantially change the distribution for this variable (particularly if the samples were biased toward relatively contaminated or uncontaminated areas).

- Our analysis focussed on fairly small carnivorous wading birds that forage exclusively in the AOCs and the reference areas. Many carnivorous wading bird species, however, are larger and forage over broader areas. We would expect risks to these bird species to be lower than for the hypothetical receptors considered in our analyses.

- The effects analyses pointed out several key sources of uncertainty. First, no data were available for any carnivorous wading bird species. Second, differing environmental conditions between the laboratory and the field introduced uncertainty to the estimation of effects doses.

The above described limitations are common to wildlife risk assessments and indicate the value of having other lines of evidence to help characterize risks. Biological surveys and ambient toxicity testing are two such lines of evidence. No *in situ* or whole media feeding studies are available, however, for carnivorous wading birds in the Calcasieu Estuary. Formal biological surveys that relate degree of COC contamination to abundances of different carnivorous wading bird species have not been conducted. However, bird banding and other surveys indicate that many species of carnivorous wading birds are common throughout the estuary. While this evidence certainly cannot be used to rule out the possibility that COCs are causing adverse effects to carnivorous wading birds in the AOCs, it does seem unlikely that any COCs are causing widespread impacts to carnivorous wading birds on a larger spatial scale.

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Tables

Table H2-1. Risk quotients for contaminants of concern.

Contaminant of Concern (COC)	Area	Risk Quotient	Proceed to Probabilistic Assessment?
Mercury	Bayou d'Inde	16.3	Yes
	Upper Calcasieu River	3.5	Yes
	Middle Calcasieu River	3.7	Yes
	Reference Areas	1.5	Yes
TCDD – TEQs	Bayou d'Inde	1	Yes
	Upper Calcasieu River	0.2	No
	Middle Calcasieu River	0.9	No
	Reference Areas	0.4	Yes

TCDD = tetrachlorodibenzo-*p* -dioxin; TEQs = toxic equivalents.

Table H2-2. Monte Carlo analysis input variables.

Variable			Distribution	Parameters
Body Weight: average-sized species (*BW*; kg)			normal	Mean = 1.5, SD = 0.22
Body Weight: small-sized species (*BW*; kg)			normal	Mean = 0.9, SD = 0.14
Free Metabolic Rate: average-sized and small species (*FMR*; Kcal/kg bw/day)			FMR = aBW^b	
a = FMR-slope			normal	Mean = 0.681, SD = 0.102
b = FMR-power			normal	Mean = 0.749, SD = 0.037
Gross Energy (GE_f; Kcal/kg)			lognormal	Mean = 1200, SD = 240
Assimilation Efficiency (AE_f; Unitless)			beta	alpha = 20, beta = 6.5, scale = 1.0
Contaminants of Concern (COCs) - Input for Monte Carlo				
COCs	**Area**	**Tissue Classification**		
Mercury	Bayou d'Inde	C_{fish} (mg/kg ww)	lognormal	Mean = 0.13, SD = 0.005
	Middle Calcasieu River	C_{fish} (mg/kg ww)	lognormal	Mean = 0.051, SD = 0.001
	Upper Calcasieu River	C_{fish} (mg/kg ww)	lognormal	Mean = 0.041, SD = 0.001
	Reference Areas	C_{fish} (mg/kg ww)	lognormal	Mean = 0.023, SD = 0.0001
TCDD − TEQs	Bayou d'Inde	C_{fish} (ng/kg ww)	lognormal	Mean = 64.8, SD = 1.15
	Reference Areas	C_{fish} (ng/kg ww)	lognormal	Mean = 19.1, SD = 0.21

SD = Standard deviation; NA = Not Applicable; TCDD = tetrachlorodibenzo-*p*-dioxin; TEQs = toxic equivalents.

Table H2-3. Probability Bounds analysis input variables.

Variable			Distribution	Parameters
Body Weight: average-sized species (*BW* ; kg)			normal	Mean = 1.5, SD = 0.22
Body Weight: small-sized species (*BW* ; kg)			normal	Mean = 0.9, SD = 0.14
Free Metabolic Rate: average-sized and small species (*FMR* ; Kcal/kg bw/day)			$FMR = aBW^b$	
a = FMR-slope			normal	Mean = 0.681, SD = 0.102
b = FMR-power			normal	Mean = 0.749, SD = 0.037
Gross Energy (GE_f; Kcal/kg)			lognormal	Mean = 1200, SD = 240
Assimilation Efficiency (AE_f, unitless)			minmaxmean	0.46, 1.00, 0.76
Contaminants of Concern (COCs) - Input for Probability Bounds				
COCs	**Area**	**Tissue Classification**	**Distribution**	**Parameters**
Mercury	Bayou d'Inde	C_{fish} (mg/kg ww)	lognormal	Mean = 0.065, SD = 0.0063
	Middle Calcasieu River	C_{fish} (mg/kg ww)	lognormal	Mean = 0.178, SD = 0.0178
	Upper Calcasieu River	C_{fish} (mg/kg ww)	lognormal	Mean = 0.0534, SD = 0.0056
	Reference Areas	C_{fish} (mg/kg ww)	lognormal	Mean = 0.0282, SD = 0.0035
TCDD – TEQs	Bayou d'Inde	C_{fish} (ng/kg ww)	uniform	Min = 20.5; Max=569
	Reference Areas	C_{fish} (ng/kg ww)	uniform	Min = 6.65; Max=64.8

SD = Standard deviation; NA = Not Applicable; TCDD = tetrachlorodibenzo-*p*-dioxin; TEQs = toxic equivalents.

Table H2-4. Summary of exceedance probabilities for carnivorus wading birds from Calcasieu Estuary.

Location	Probability of Exceedance (%)											
	Average-Sized Carnivorous Wading Birds						Small Carnivorous Wading Birds					
	LB		*FOMC*		*UB*		*LB*		*FOMC*		*UB*	
	LT	UT	LT	UT	LT	UT	LT	UT	LT	UT	LT	UT
Mercury												
Bayou d'Inde	0	0	0	0	82	0	0	0	0	0	91	0
Middle Calcasieu River	0	0	0	0	11	0	0	0	0	0	15	0
Upper Calcasieu River	0	0	0	0	7	0	0	0	0	0	10	0
Reference Areas	0	0	0	0	0	0	0	0	0	0	0	0
TCDD − TEQs												
Bayou d'Inde	0	0	0	0	100	11	0	0	7.4	0	100	8
Reference Areas	0	0	0	0	96	0	0	0	0	0	99	0

LB = Lower Probability Bound; FOMC = First Order Monte Carlo; UB = Upper Probability Bound; LT = Lower Toxicity Threshold; Upper Toxicity Threshold;
TCDD = tetrachlorodibenzo-*p* -dioxin; TEQs = toxic equivalents.

Table H2-5. Summary of exceedance probabilities for the Appendix G benchmarks.

Location	Benchmark	Probability of Exceedance (%)					
		Average-Sized Carnivorous Wading Birds			Small Carnivorous Wading Birds		
	Benchmark	*LB*	*FOMC*	*UB*	*LB*	*FOMC*	*UB*
Mercury							
Bayou d'Inde	0.0202 mg/kg bw/d	0	0	100	0	0	100
Middle Calcasieu River		0	0	89	0	0	95
Upper Calcasieu River		0	0	81	0	0	89
Reference Areas		0	0	32	0	0	40
TCDD -- TEQs							
Bayou d'Inde	44.3 ng/kg bw/d	0	0	100	0	0	100
Reference Areas		0	0	18	0	0	24

LB = Lower Probability Bound; FOMC = First Order Monte Carlo; UB = Upper Probability Bound; TCDD = tetrachlorodibenzo-*p* -dioxin; TEQs = toxic equivalents.

Figures

Figure H2-1. Overview of approach used to assess exposure of carnivorous wading birds to contaminants of concern (COCs) in the Calcasieu Estuary.

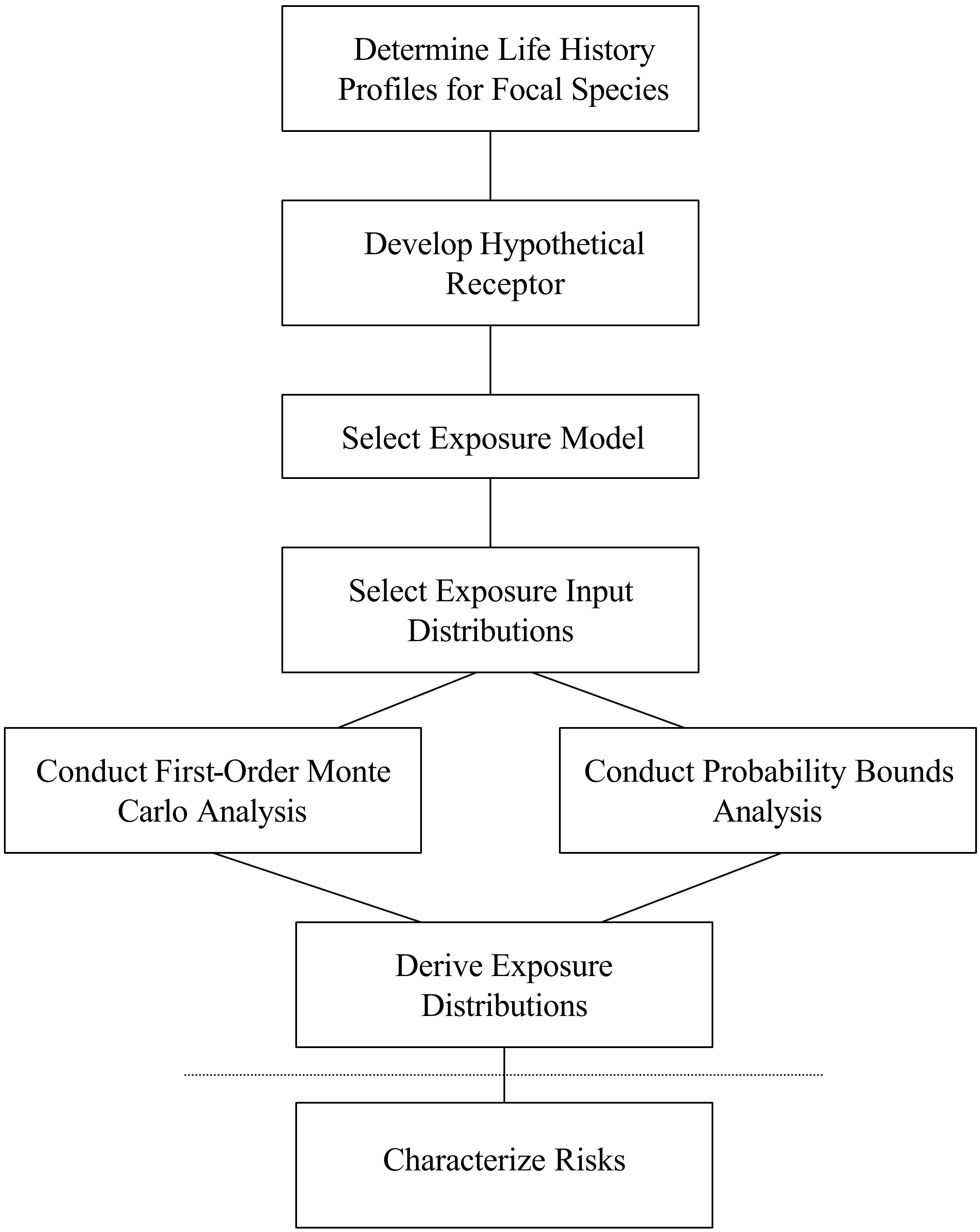

Figure H2-2. Overview of approach used to assess the effects of carnivorous wading birds exposed to contaminants of concern (COCs) in the Calcasieu Estuary.

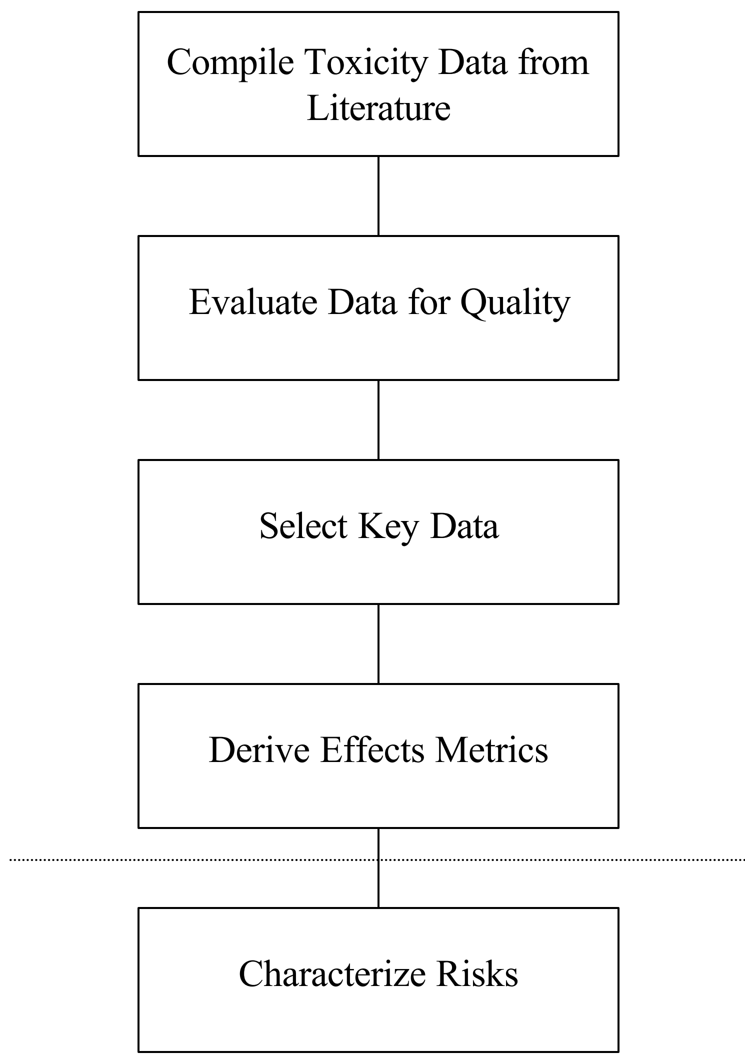

Figure H2-3. Overview of approach used to assess the risks of carnivorous wading birds exposed to contaminants of concern (COCs) in the Calcasieu Estuary.

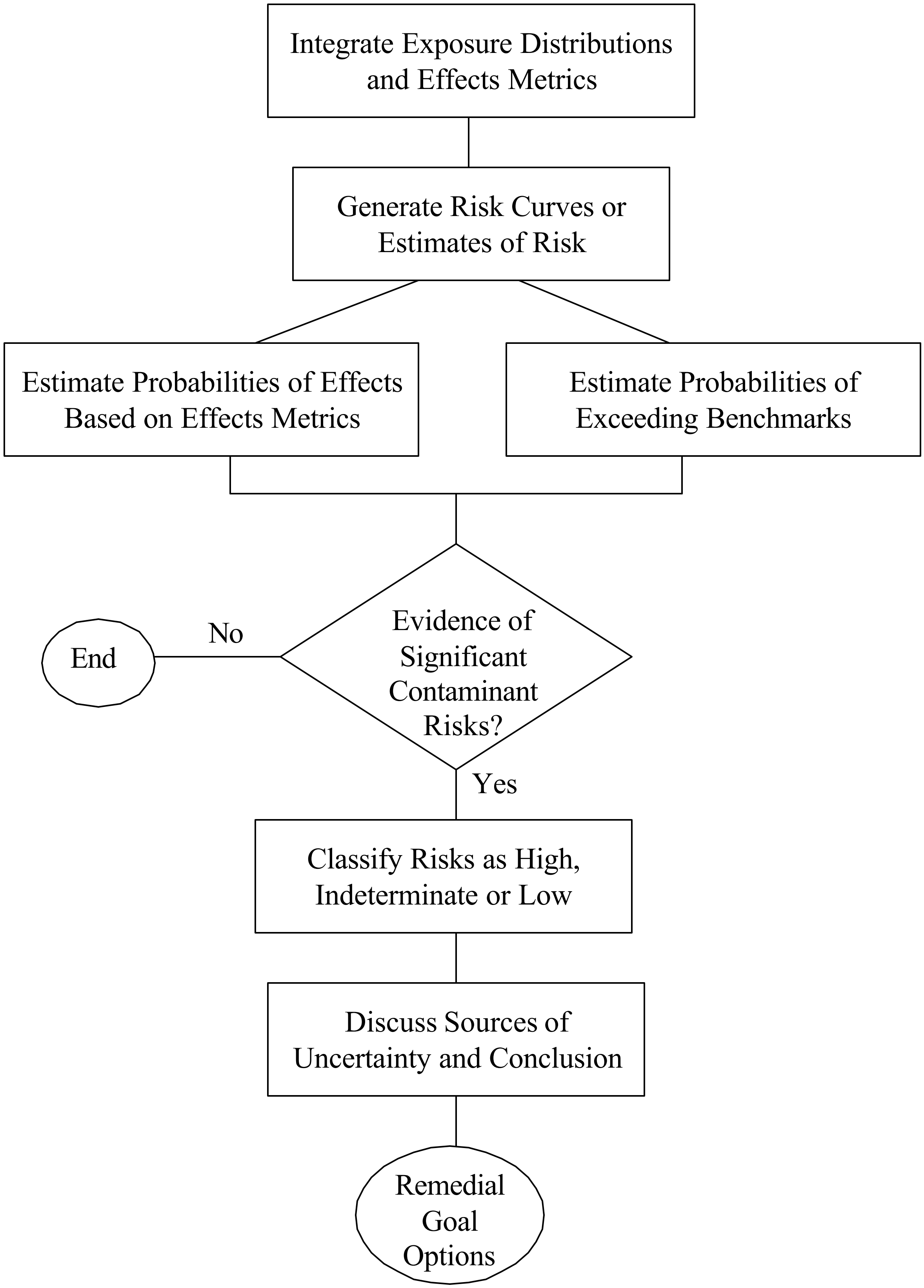

Figure H2-4. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized carnivorous wading birds in Bayou d'Inde, Calcasieu Estuary.

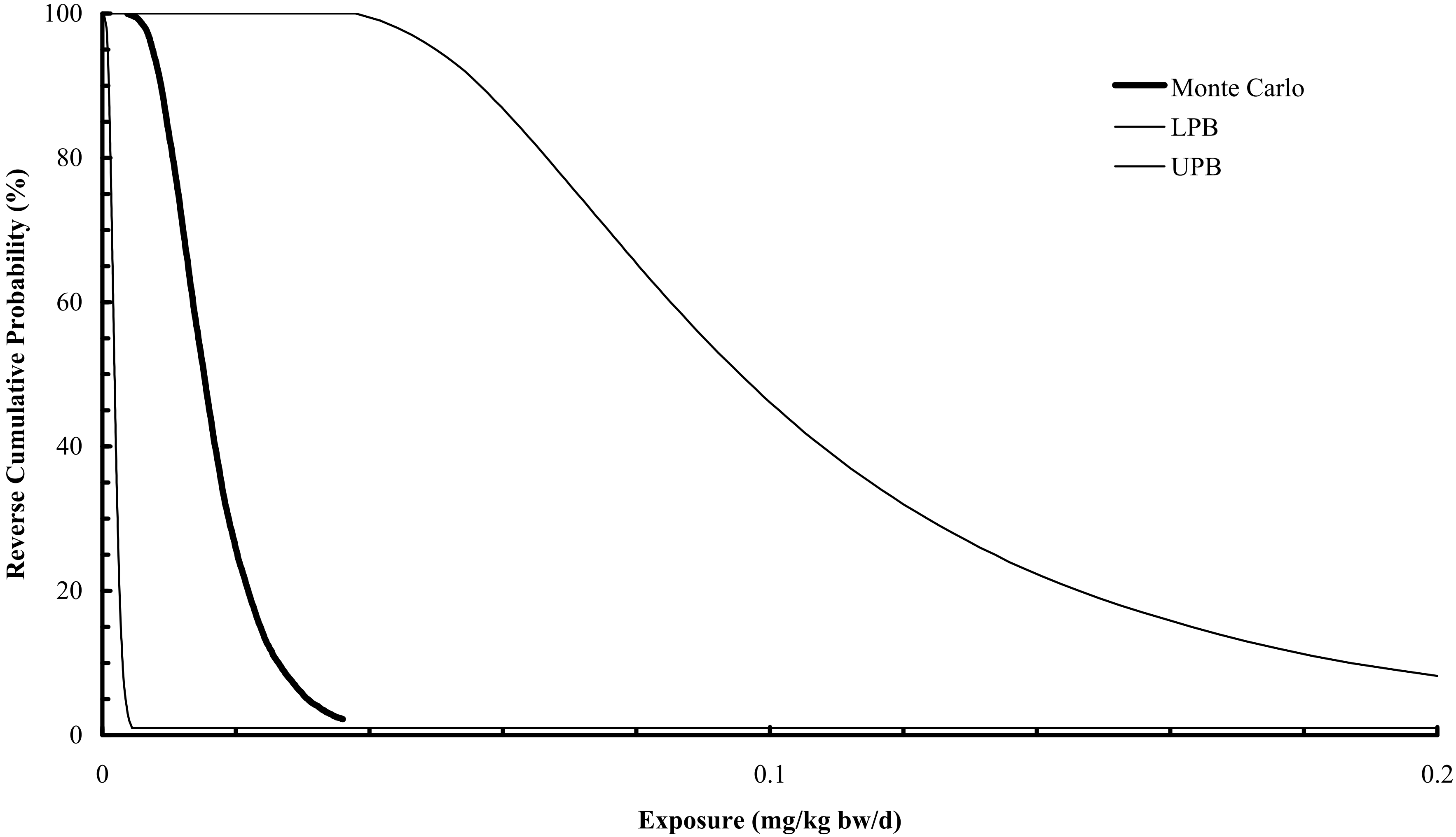

Figure H2-5. Reverse cumulative probability distribution of total daily intake rates of mercury by small carnivorous wading birds in Bayou d'Inde, Calcasieu Estuary.

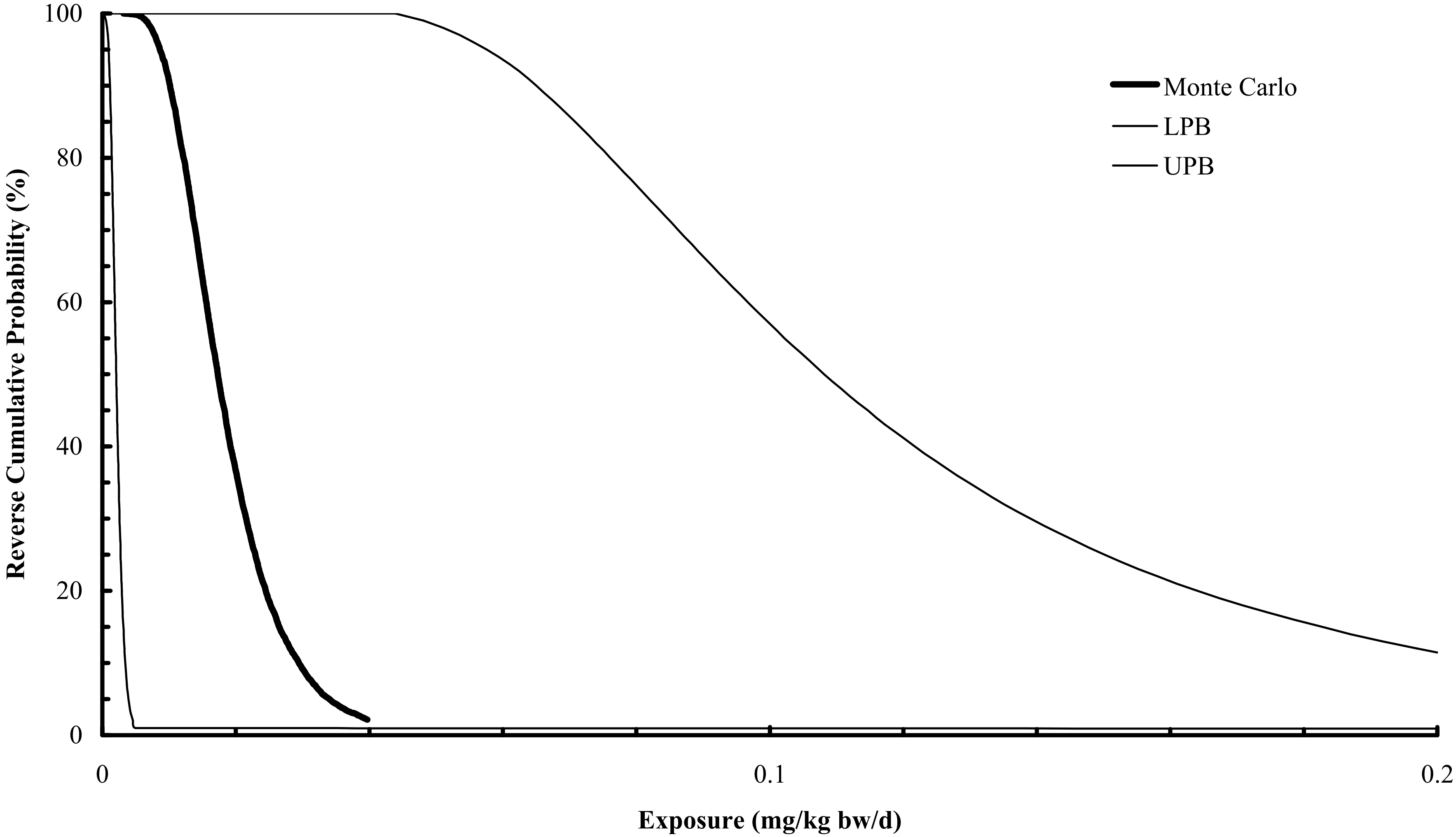

Figure H2-6. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized carnivorous wading birds in the Middle Calcasieu River, Calcasieu Estuary.

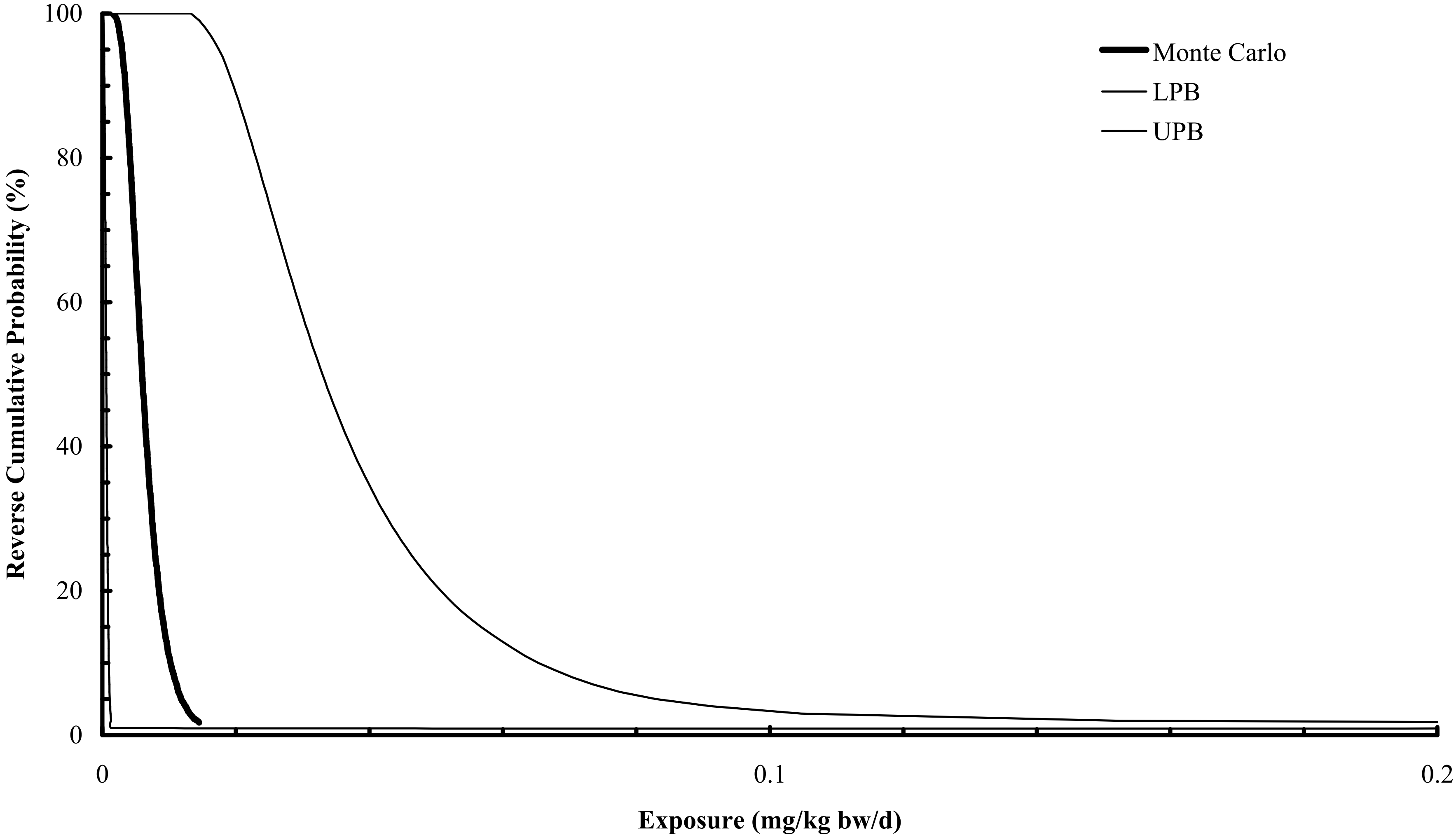

Figure H2-7. Reverse cumulative probability distribution of total daily intake rates of mercury by small carnivorous wading birds in the Middle Calcasieu River, Calcasieu Estuary.

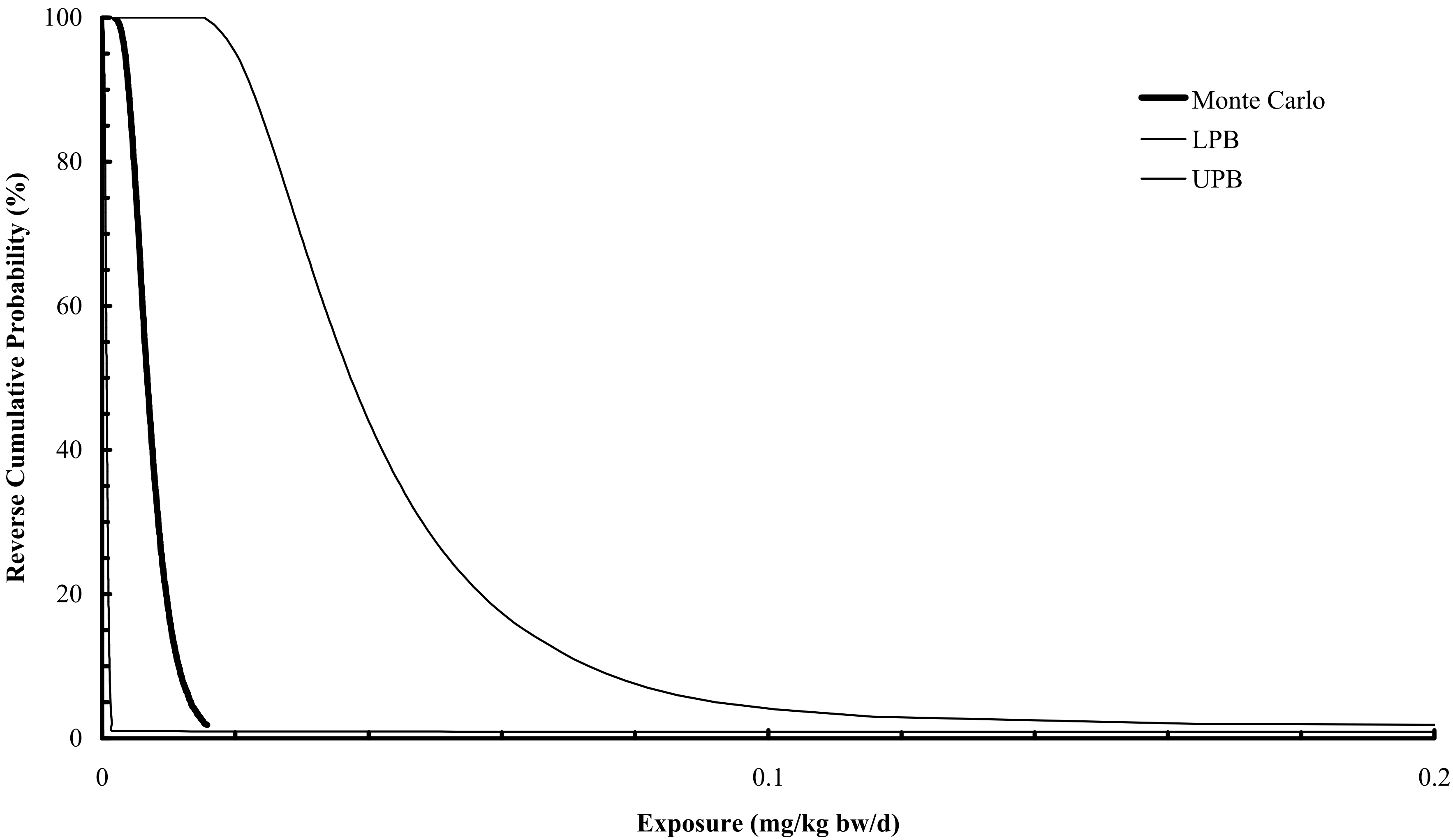

Figure H2-8. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized carnivorous wading birds in the Upper Calcasieu River, Calcasieu Estuary.

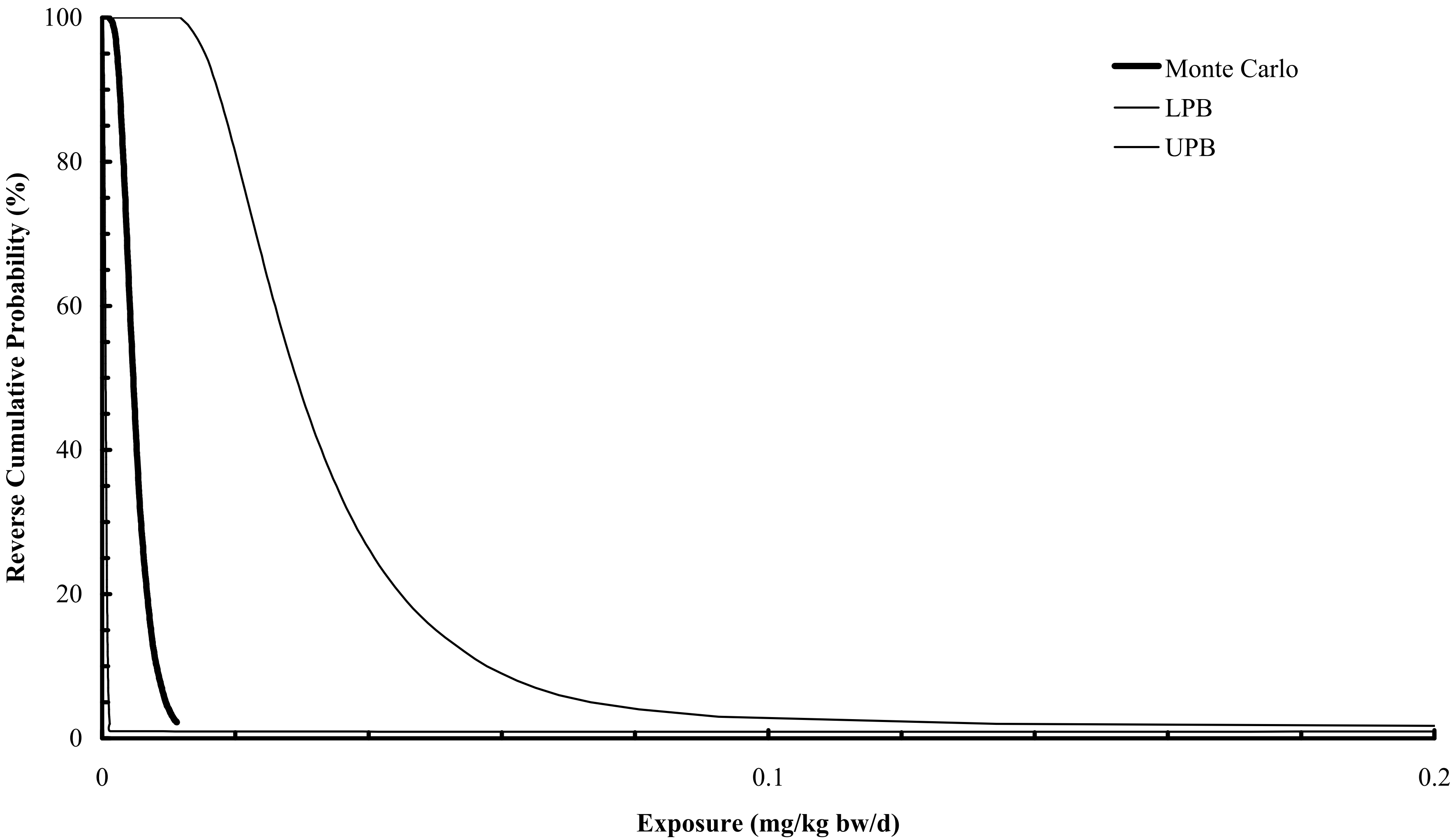

Figure H2-9. Reverse cumulative probability distribution of total daily intake rates of mercury by small carnivorous wading birds in the Upper Calcasieu River, Calcasieu Estuary.

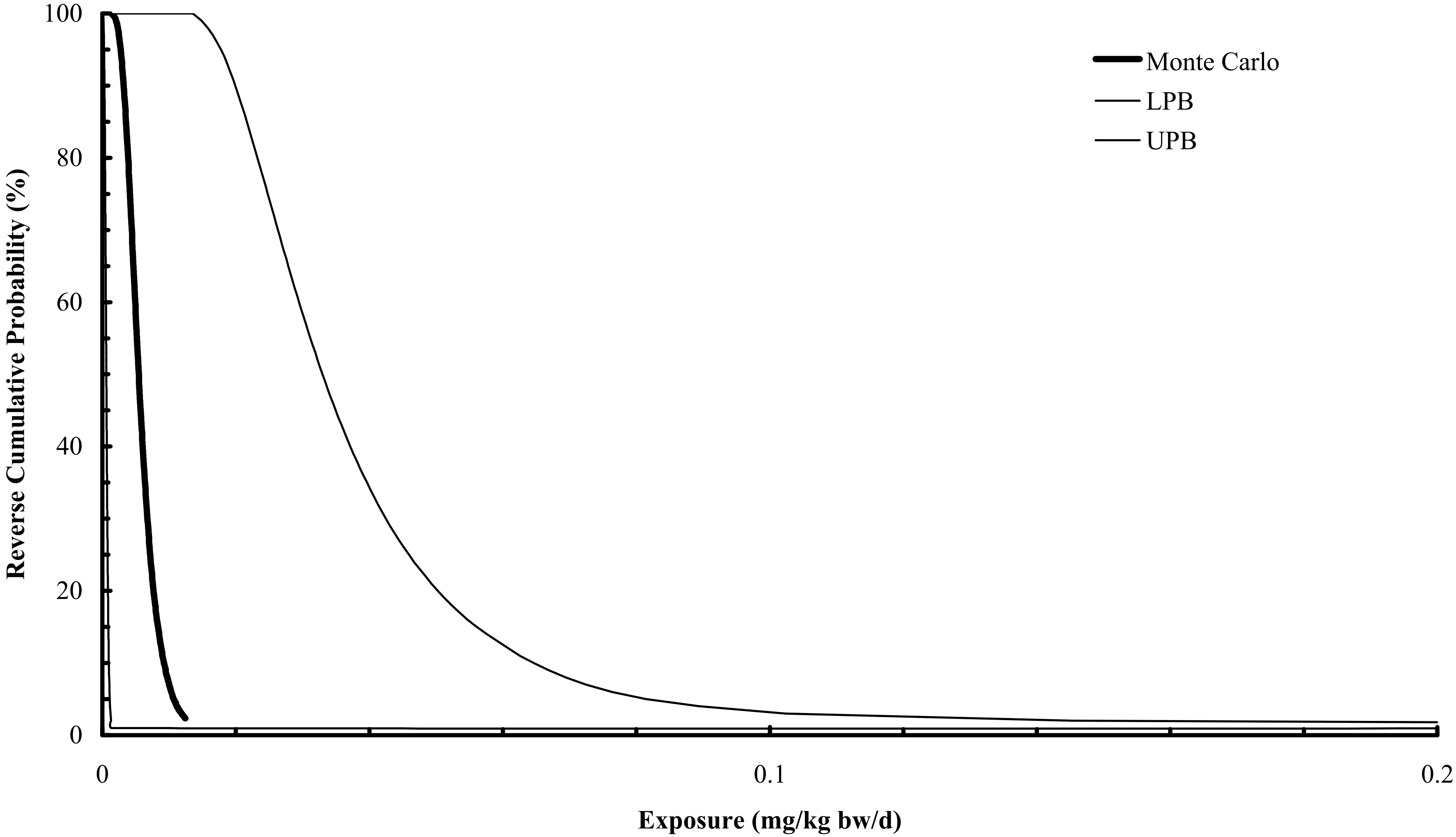

Figure H2-10. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized carnivorous wading birds in the reference areas, Calcasieu Estuary.

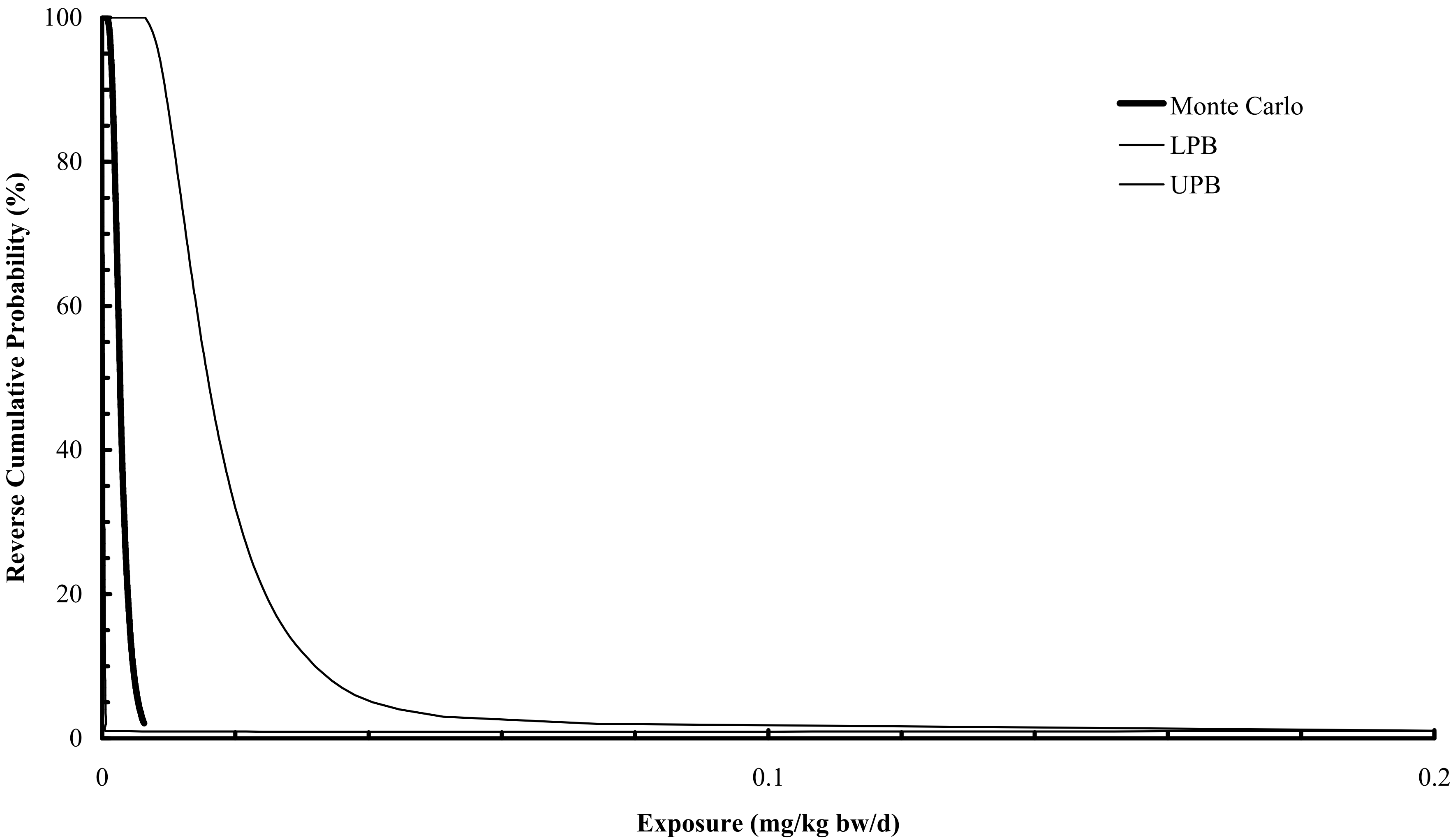

Figure H2-11. Reverse cumulative probability distribution of total daily intake rates of mercury by small carnivorous wading birds in the reference areas, Calcasieu Estuary.

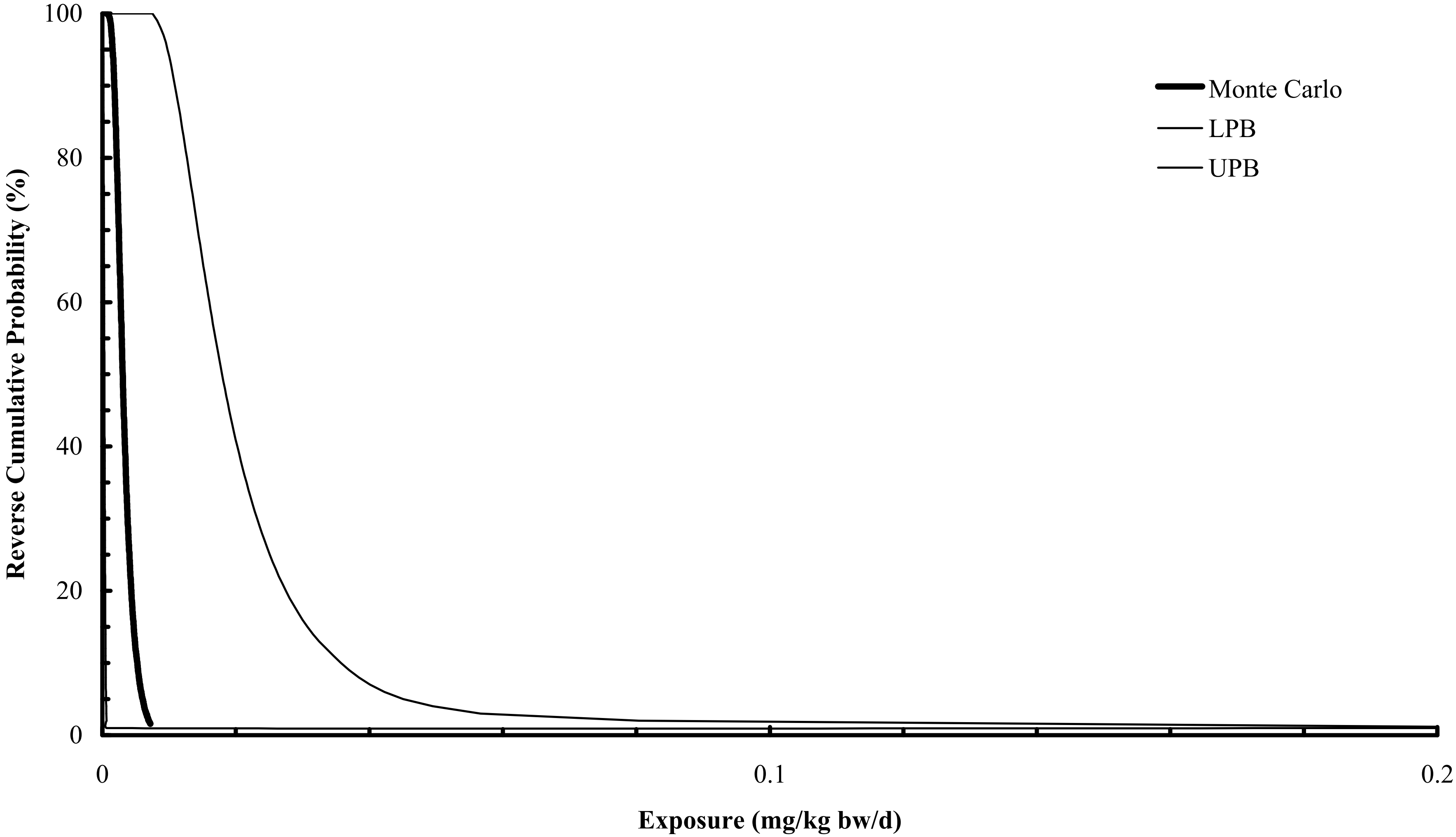

Figure H2-12. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by average-sized carnivorous wading birds in Bayou d'Inde, Calcasieu Estuary.

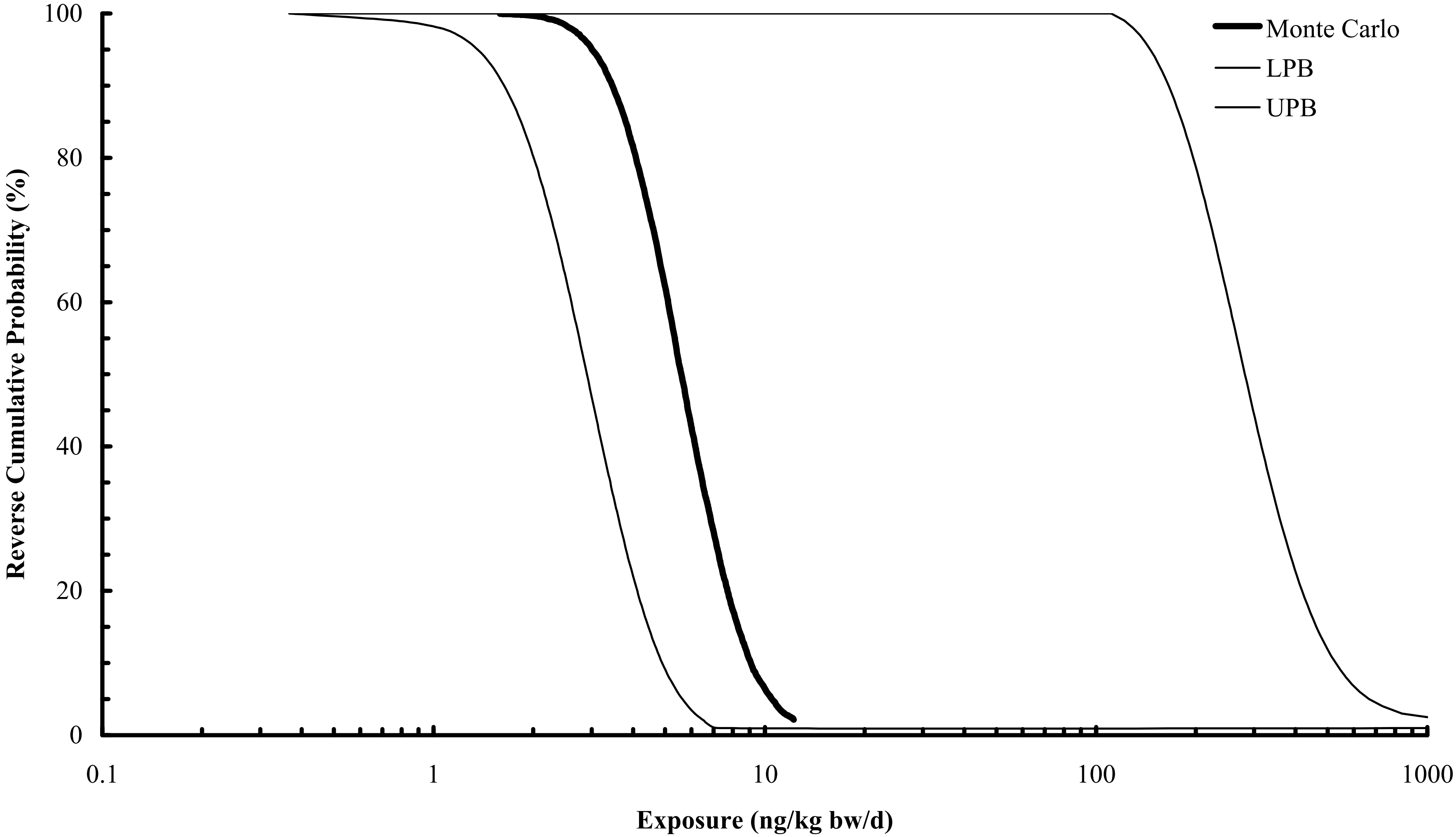

Figure H2-13. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by small carnivorous wading birds in Bayou d'Inde, Calcasieu Estuary.

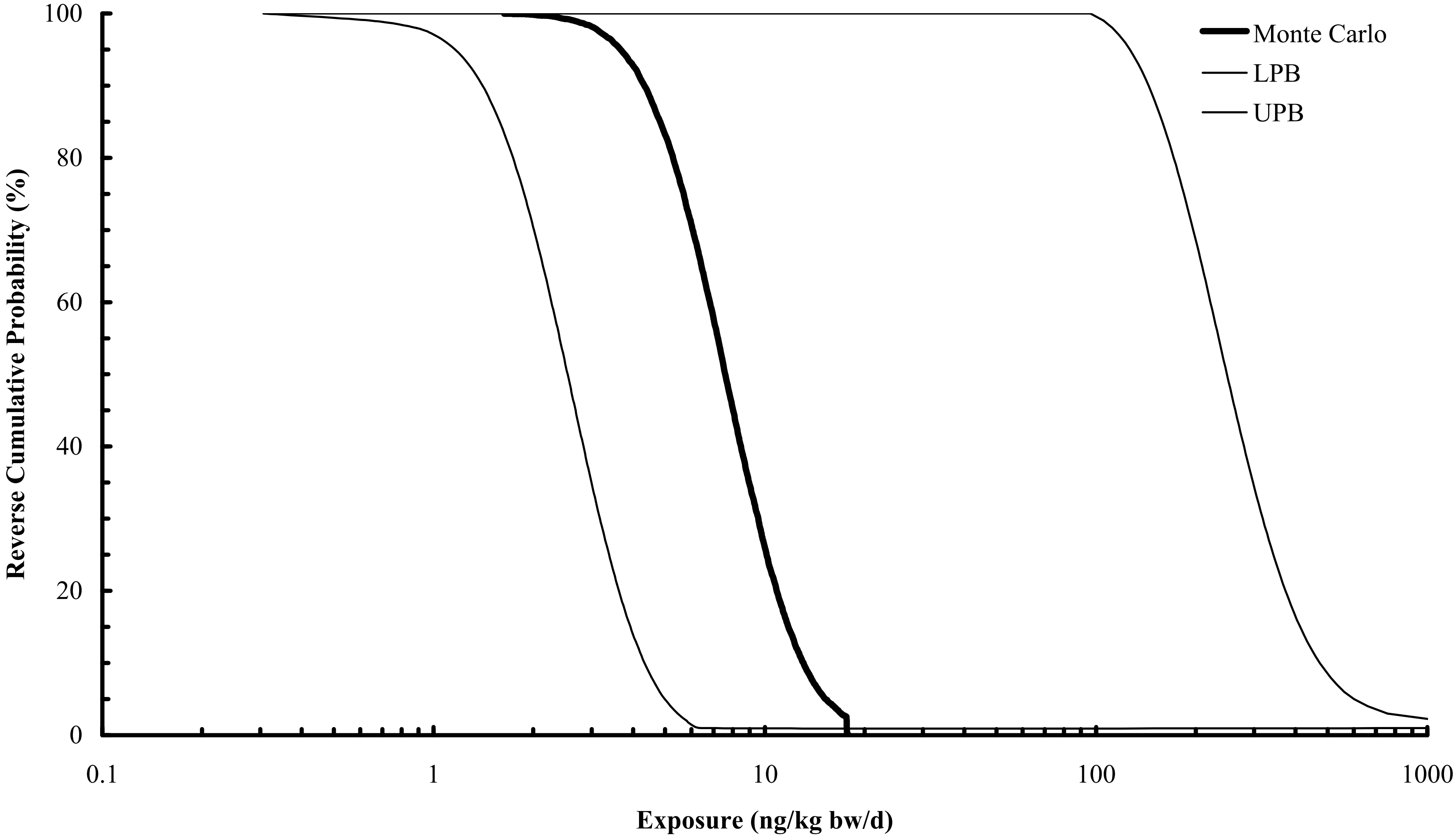

Figure H2-14. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by average-sized carnivorous wading birds in the reference areas, Calcasieu Estuary.

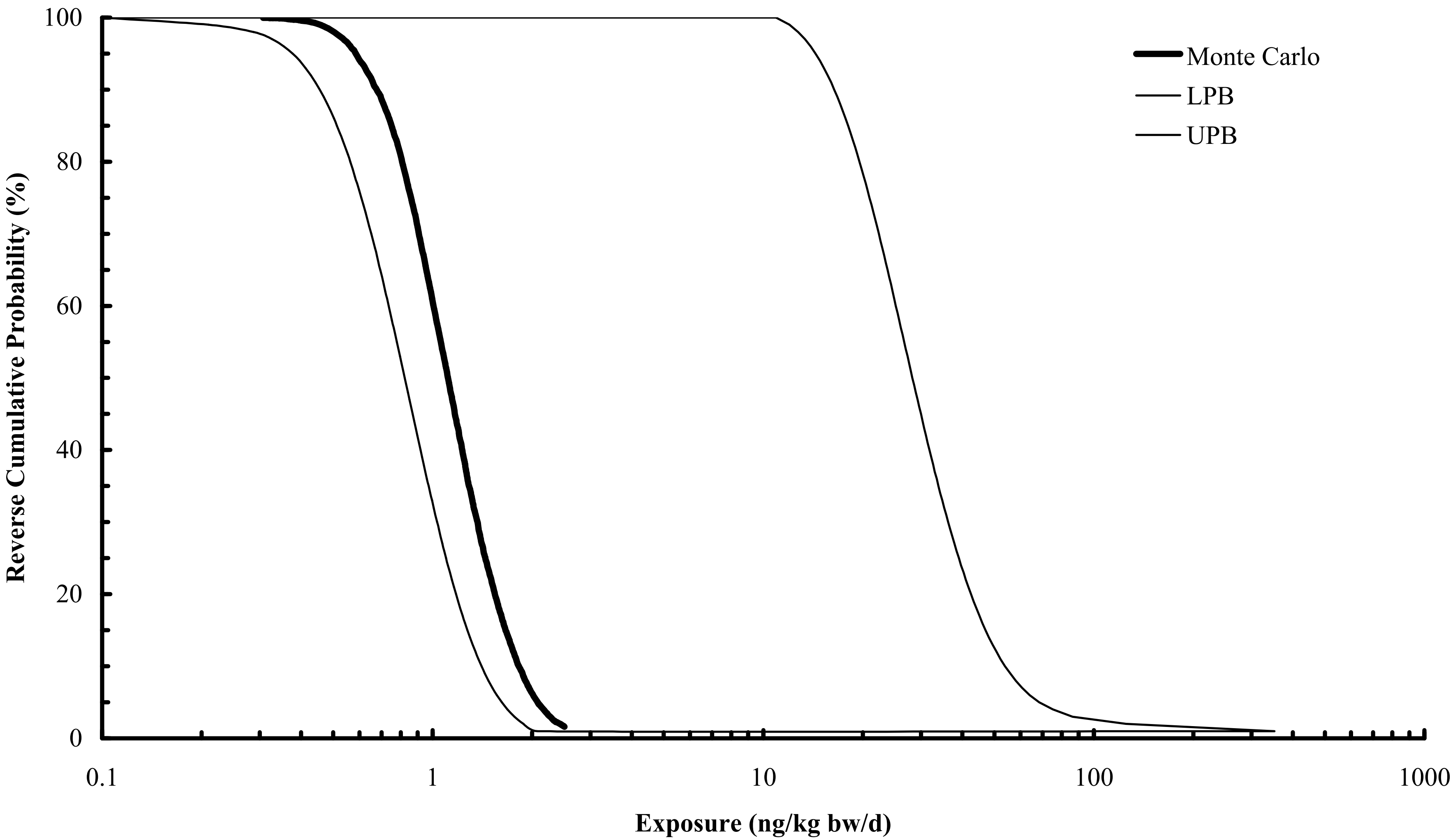

Figure H2-15. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by small carnivorous wading birds in the reference areas, Calcasieu Estuary.